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Multi-species Management Using Modeling and Decision Theory
Applications to Integrated Natural Resources Management Planning

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MULTI-SPECIES MANAGEMENT USING MODELING AND DECISION THEORY:

APPLICATIONS TO INTEGRATED NATURAL RESOURCES MANAGEMENT PLANNING

June 2008

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Cover Photo Credits *– from top left 1) Engelmann oak and fire (D.M. Lawson); 2) Orcutt's spineflower (D.M. Lawson); 3) Dead wart-stemmed ceanothus (D.M. Lawson); 4) San Clemente Island sage sparrow (Nicole Munkwitz) 5) Live wart-stemmed ceanothus (D.M. Lawson) 6) California Least Tern (Steven Eric Smith)*

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1.0 INTRODUCTION

Advances in computing power and software development have made it feasible to incorporate complicated modeling endeavors in routine natural resources management planning. In this report we review the theory and application of models and decision theory and make recommendations for their use in the context of the Department of Defense (DoD). Model-based risk assessments are powerful decision tools. This is not to say that models will make hard decisions easy or replace management. They require ecological expertise to develop and interpret, and should be applied within a formal decision framework. What models can do is to clarify risks and trade-offs among choices and inform data collection. A formal approach allows for the explicit statement of assumptions, assures logical consistencies, and allows for the incorporation of methods to deal with uncertainties or knowledge gaps in calculations. Models have been used to assess impacts of known stressors, evaluate management options, and focus future data collection on the most important data gaps (Akçakaya 2000). To be most effective models must be updated as part of an adaptive management program. As information is obtained it can be fed back into the model(s) and used to evaluate existing management strategies or develop new ones. Lastly, modeling as a decision making tool has multiple potential applications to natural resources management and planning within the DoD including impact analyses required by the National Environmental Policy Act (NEPA); establishing management goals and objectives within the installation's Integrated Natural Resources Management Plan; generation of an ecosystem health metric for evaluating the efficacy of management programs; and helping to convey DoD's natural resources management accomplishments to the public.

Modeling is particularly useful in complex ecosystems where descriptive analyses of observational data alone are not powerful enough to examine the interactions and trade-offs among organisms. The use of metapopulation models can help define which subpopulations on installations are most important for species persistence, facilitating analysis of trade-offs between military training and conservation. In species with metapopulation structures, trade-offs could be made where military training is allowed in some subpopulations and restricted in others depending on the value of the different areas to training and species persistence.

■ *Modeling is particularly useful in complex ecosystems where descriptive analyses of observational data alone are not powerful enough to examine the interactions and trade-offs among organisms.*

The loggerhead sea turtle offers a powerful early example of the success of a formal model-based approach. Until the 1980s sea turtle management focused on protecting nesting sites and hatchlings as they dispersed to the sea after emerging from their eggs. However, this didn't result in increases in nesting turtles. A model-based approach revealed that population growth rates were particularly sensitive to sub-adult mortality and much less sensitive to hatchling mortality (Crouse et al. 1987). This eventually led to widespread use of turtle exclusion devices by the fishing

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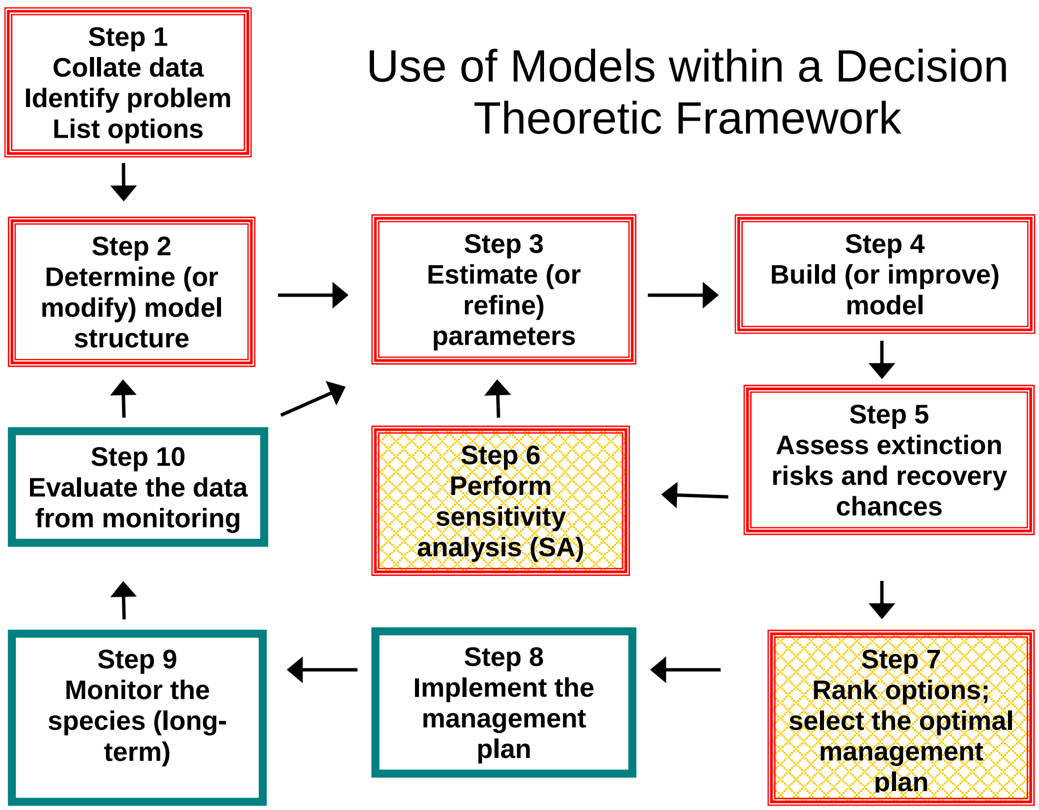
industry and subsequent increases in the turtle population (Crowder et al. 1994).

Within the DoD are numerous potential applications of model based decision making that would support natural resources managers and the military mission. The key is to use models as decision support tools within a formal decision framework rather than basing decisions solely on model output (Burgman 2005). Figure 1 shows how models can be incorporated in a decision theoretic framework.

Figure 1. Decision Theoretic Framework. Red boxes represent the planning and analysis phase. Blue boxes represent plan implementation and initiation of adaptive management. Steps six and seven generate management tasks (modified from Akçakaya et al. 1999).



The framework (Figure 1) can be separated into two components. The boxes outlined in red (steps one through seven) encompass the analysis and planning phase. The second phase, outlined in blue (steps eight through ten), involves implementation of the plan and initiation of adaptive management; the results of this second phase feed back into analysis and plan revision on a specified schedule. Steps 6 and 7 represent points where tasks can be generated for Integrated Natural Resources Management Plans (INRMPs). Impacts can be assessed at step 8. Once the known information has been assembled the models can be run using

various impact scenarios to evaluate relative impacts of the proposed activities and alternatives. Cumulative impacts can be assessed in a similar manner by incorporating effects from projects in a specified region and time frame into analyses run at step eight.

While in an ideal world the above schematic would be followed and decisions would be made after the appropriate information was collected and analyzed, many decisions are made with existing information or information that can be generated within a relatively short time frame such as a year. Fortunately, this framework can be used even if the only information available initially is expert opinion. In that case a model could be developed with expert opinion and steps six and seven generate the tasks to be specified in the plan including management actions, and data collection for model parameter refinement. Data to evaluate the chosen management strategy is generated in step nine. Step 10, evaluation of the monitoring data, provides feedback both on the accuracy model and the success of the chosen management action. Adaptive management is implemented by subsequent model revision and re-evaluation of selected management options.

The use of transparent formal decision-making methods allows stakeholders and the public to see the relative importance of different assumptions used in the decision making process and can assist in evaluating alternate scenarios. This flexibility and transparency can be used to increase public confidence in DoD natural resource decision making. Because the analysis is transparent and repeatable, conclusions can more easily be communicated and defended to regulators and the public.

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While a formal approach clarifies choices, when the uncertainty is fully accounted it may still not be unambiguous which choice is the best. In those cases, other considerations may have to be taken into account such as availability of resources to implement management, and opportunity costs to other programs. Also, at times decisions have to be made relatively quickly and managers may not have the time to develop and implement a formal model, at least not initially. Emergencies are obvious examples of decisions that cannot be deferred. In other cases, the political will may not exist to defer a decision because of insufficient data. For example, while society may draw the line at taking an action that knowingly jeopardizes the survival of an endangered species, society may be unwilling to defer a decision until sufficient data exist to have confidence that the chosen course will ensure survival. The Federal Endangered Species Act (ESA) requires the use of the best available data with limited flexibility to postpone decisions without mutual agreement from all parties (USFWS and NMFS 1998). In the case of insufficient data, the "benefit of the doubt" is to be given to the species. Specific guidance on how to do this, however, is not given and when the

uncertainty is high it may be difficult to make meaningful decisions. One obvious solution is to proceed using an adaptive management approach that implements management strategies using an experimental framework to develop improved methods.

- *Not using models doesn't make the uncertainty less or the resultant decisions more robust but it can obscure the fact that the uncertainty exists, giving false confidence in decisions made concerning those species.*

The use of models for decision making is not without its detractors; criticisms of model-based decision-making are predominantly based on lack of model accuracy. Critics point out that when so much uncertainty surrounds parameters used in the models the resulting error renders predictions unreliable or fails to produce clear choices between alternatives. While this certainly presents problems, not using models in those cases doesn't make the uncertainty less or the resultant decisions more robust but it can obscure the fact that the uncertainty exists, giving false confidence in decisions made concerning those species.

2.0 BACKGROUND

DoD natural resource managers regularly face compelling, competing demands for conservation funds, and military mission land use needs can mean tough choices about where and how natural resources are managed. Natural resources are managed on DoD's installations through INRMPs. The Sikes Act Improvement Act (16 USC 670 *et. seq.*) requires the preparation of INRMPs and specifies that the military mission be integrated and balanced with natural resources management resulting in no net loss of military training as a result of plan implementation. DoD instruction 4715.3 requires an ecosystem approach to natural resources management. In addition to the Sikes Act, the DoD must comply with the ESA which requires that agency actions not jeopardize the continued existence of species listed under the Act and directs agencies to use their authority to conserve those species. The management of rare and endangered species involves more than just compliance with statutory legal requirements; it also calls for managing and planning for species that are likely to be added to the lists to either forestall their listings or to avoid surprises when they are listed. Individual species have traditionally been the focus of conservation management because of the difficulty of considering multiple species simultaneously and the imperative of intensively managing at-risk species to avoid extinction. This approach however has problems.

Single species management operates under the typically unsubstantiated assumption that other species sharing the same habitat benefit from management of the target species. This assumption has been questioned for lack of evidence and the approach criticized for directing the bulk of conservation funding to a few species while under funding other species and fundamental ecosystem management work such as soil erosion and invasive species control (Franklin 1993; Lambeck 1997). Until recently the complexities of managing multiple species simultaneously with differing population dynamics and habitat requirements in the face of

uncertain ecosystem drivers (rainfall, wildfires, etc.) depended on the skills and expertise of individual managers. Systematic multi-species management has been beyond the scope of the tools routinely implemented by conservation managers. However, recent shifts in conservation planning methodology for multiple species have resulted in the adoption and development of interdisciplinary tools to answer questions at the science-policy interface (Nicholson and Possingham 2007). In addition, as species have been added to the threatened and endangered species list, DoD installations have found themselves with the highest density of listed species of any federal landowner (Stein et al. 2008). With small and overlapping populations of multiple listed species DoD cannot effectively optimize both its mission and endangered species conservation using conventional single species approaches.

■ *As species have been added to the threatened and endangered species list, DoD installations have found themselves with the highest density of listed species of any federal landowner (Stein et al. 2008).*

Recent ESA exemptions from designation of critical habitat granted to the DoD rely on management provided through the INRMP process to ensure conservation of listed species. The Government Accounting Office (GAO 2003) concluded that some of the measures necessary to conserve listed species restrict military land uses and potentially compromise DoD's mission. In order to provide for conservation but reduce opportunity costs to military training of endangered species conservation, the 2004 National Defense Authorization Act qualifies INRMPs as "special management plans". This designation means that if the plans provide effective conservation benefits to the specific species and ensure that the conservation actions will be implemented, critical habitat for the covered species will not be designated under the ESA on that installation. Environmental groups are concerned that without critical habitat restrictions, DoD's conservation efforts will be inadequate. To complicate this, significant benefits from critical habitat designation have not been documented (Hoekstra et al. 2002) nor have there been definitive studies performed to demonstrate the effectiveness of INRMPs in conserving endangered species. DoD has an interest in demonstrating more clearly that additional regulation is unnecessary and that implementation of INRMPs is more cost-effective, feasible and biologically relevant than managing species through critical habitat designation.

In addition to requirements to manage natural resources, both the ESA and NEPA require assessment and compensation for direct, indirect and cumulative effects to threatened and endangered species and other natural resources. Assessment of cumulative impacts is complex and NEPA documents typically focus on short-term impacts. Long-term, indirect and cumulative impacts, however, often pose the most significant threats (Wheeler et al. 2005) and are those most difficult to accurately predict. Methods to assess and aggregate impacts from multiple projects or sources within a few projects are needed. Natural resources management as well as impact assessment and the associated identification of avoidance, minimization and compensation measures would benefit from an

improved ability to project a population's long-term response to impacts and management actions. In addition, formal treatment of the interactions or trade-offs among organisms and between organisms and environmental factors should lead to an increased ability to predict impacts and the outcomes of management actions.

- *With diminishing funding for natural resource programs it is critical to obtain maximum benefit from available conservation management budgets.*

With diminishing funding for natural resource programs it is critical to obtain maximum benefit from available conservation management budgets. The best outcome for a natural resource management plan is to meet multiple management goals with each strategy, rather than piecing together a range of, sometimes conflicting, focused management strategies for single species. One way to improve the outcome of management choices is to consider multiple species simultaneously. We propose methods to use population models and formal decision making methods to do so.

- *The military mission is incorporated in two ways. The first is in crafting management alternatives for analysis that are compatible with the mission and the second is in the decision framework that explicitly lays out the trade-offs between mission opportunities, species persistence and cost.*

The first component of the approach proposed here is to construct population models, based on current knowledge, to quantitatively evaluate the effects of alternative management strategies on a range of locally at-risk species. The results of the modeling component feed into a decision-theoretic framework to optimize decisions under multiple objectives and competing goals. This framework will be based on assessing goals, objectives and values of individual stakeholders and the decision maker and then selecting the optimal management strategy. The ultimate decision framework will assist in developing cost-effective management strategies that optimize the persistence of the targeted species while maintaining the military mission. The military mission is incorporated in two ways. The first is in crafting management alternatives for analysis that are compatible with the mission, and the second is in the decision framework that explicitly lays out the trade-offs between mission opportunities, species persistence and cost.

This proposed framework can assist the military in shifting from single-species to multiple species management by setting up formal, transparent and repeatable methods to do so. It can be used to highlight the trade-offs between conservation targets and between conservation targets and military land uses. This approach can incorporate and augment established spatial decision support tools such as NatureServe (NatureServe 2005). NatureServe consists of network of independent natural heritage programs for much of the Western Hemisphere. These programs maintain extensive databases of rare and endangered species and ecological communities including spatial locations and data on status and threats. The DoD works with NatureServe to develop and analyze information. Spatial systems such as NatureServe look at the extent of species habitats across the landscape and can address issues relating to habitat size and configuration. For example one current NatureServe project for DoD is an analysis of locations of rare species, not yet listed

under the ESA, with respect to DoD lands (NatureServe 2005). Use of this geographic analysis to prioritize species for conservation actions is intended to help stabilize populations and reduce the need for future ESA listings. Our approach takes spatial decision support systems one step further by integrating population dynamics into the decision process. The integration of population dynamics into an established spatial approach to conservation management will enable decision making that takes species life history attributes into account and can detect more subtle effects than a spatial system alone. In the example above, population dynamical information could help refine the selection to species or populations that are more at risk or more likely to benefit from a given conservation action. The methods proposed here complements spatial approaches by adding an extra dimension to the analysis.

■ *Our approach takes spatial decision support systems one step further by integrating population dynamics into the decision process.*

3.0 POPULATION VIABILITY ANALYSIS (PVA)

Most population dynamical models (Figure 2) in use in conservation can be included under the broad heading of PVA. Such models have been in use since the 1980's (Shaffer 1990), and includes methods that combine expert opinion, species specific data, formal models and risk analysis to project future population trajectories and predict probabilities of decline or extinction (Akçakaya 2004; Akçakaya 2000; Beissinger 2002; Boyce 1992; Shaffer 1990). PVA can incorporate a wide variety of information including genetic, demographic and ecological data as well as expert opinion (Akçakaya 2004; Boyce 1992).

Figure 3 is a schematic of a vital rate matrix used in a stage based model. Stages rather than ages are used for species where vital rates are dependent on size or physiological stage rather than age. For example in many woody plant species seed production and survival are functions of size rather than age. In a stochastic stage based model two matrices are included for each set of vital rates; one includes the means and the other includes the standard deviations of those means. Other types of population dynamical models are discussed under the section **Single Species Models** below.

Figure 2. Schematic representation of a stage based population dynamical model. Ovals signify significant life stages and parameters denoted by S represent survival within and between stages and by F represent fecundities.

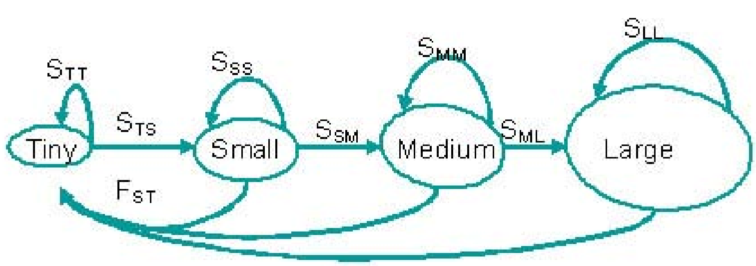
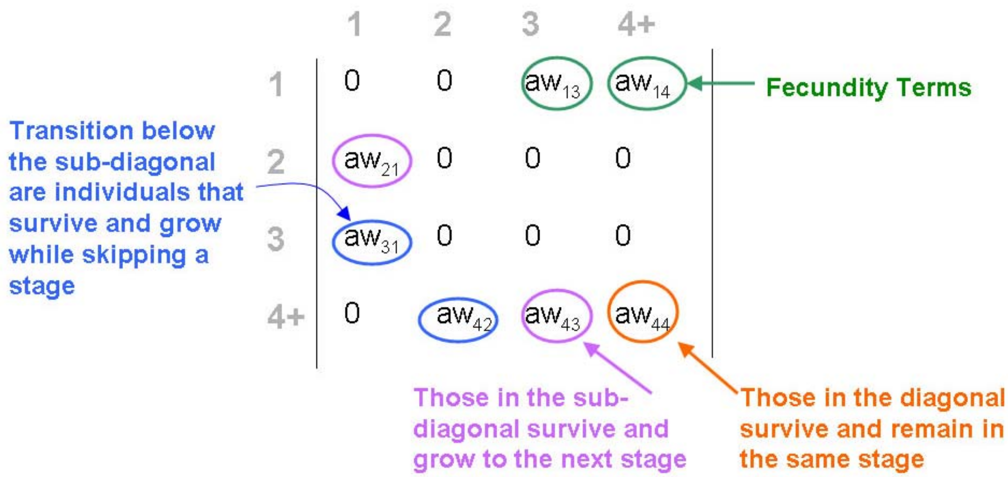


Figure 3. Vital rate matrix. The lower 3 rows include survival parameters and the top row consists of fecundities. The subscripts refer to the stages. Thus, aw21 is the survival rate for stage 1 individuals that survive and transition into stage 2.



4.0 VALIDITY OF PVAS FOR CONSERVATION:

Early in the development of PVAs it was hoped that even though data were sparse and resultant PVAs were lacking in predictive power, generalities would emerge that were useful for conservation planning (Boyce 1992; Shaffer 1990). It was believed that the Minimum Viable Population (MVP), or the population size required to ensure persistence, could be used as a general measure in conservation biology to assess populations and management options. Reed et al. (2003) concluded that this generalization may not be appropriate because of uncertainty in the estimates, and large among species variation in the estimates.

Rather than generalizations, most PVA work has remained focused on analyzing individual species. There is an ongoing debate in the scientific literature on the appropriate use of PVAs focusing on the absolute versus relative accuracy of model results and the implications thereof for specific species (McCarthy et al. 2003). A number of articles have been published since the mid-1990's evaluating the validity of PVAs. Detractors generally cite the lack of accuracy in PVA models (Coulson et al. 2001; Ellner et al. 2002; Fieberg and Ellner 2000; Ludwig 1999). Proponents, on the other hand, while generally acknowledging poor absolute accuracy, consider relative predictions sufficiently reliable to support exploratory, heuristic uses and conservation decisions (Brook et al. 2002; Lindenmayer et al. 2003; McCarthy et al. 2003; McCarthy et al. 2001). Unfortunately the lack of absolute accuracy in model estimates means that the results of evaluations of alternative management or impact scenarios are not valid for comparisons between species unless meaningful confidence intervals can be calculated. This is to say that models can only be useful in multi-species planning or impact analysis when applications will be limited to the relative rankings.

Fieberg and Ellner (2000) looked at the utility of PVAs by evaluating predictions as a function of the length of the time series length used to parameterize a PVA. They found that reliable predictions of extinction probabilities could only be made for time horizons less than 20% of the length of the data set. These time frames would have little utility in conservation planning considering that a 10 year data set could only be used to make predictions up to two years in the future. In contrast, for ranking management options, McCarthy et al. (2003) evaluated relative accuracy of PVA output using 10 years of data and found that when extinction predictions were used to select the best management option they were reliable for periods up to 100 years.

The flexibility in what constitutes a PVA is the source of some problems as it gives no assurance that all relevant data will be used. Asquith (2001) reported on the Javan gibbon and maintained that the most important factor, decline through habitat loss, was omitted from the PVA which primarily considered genetic factors.

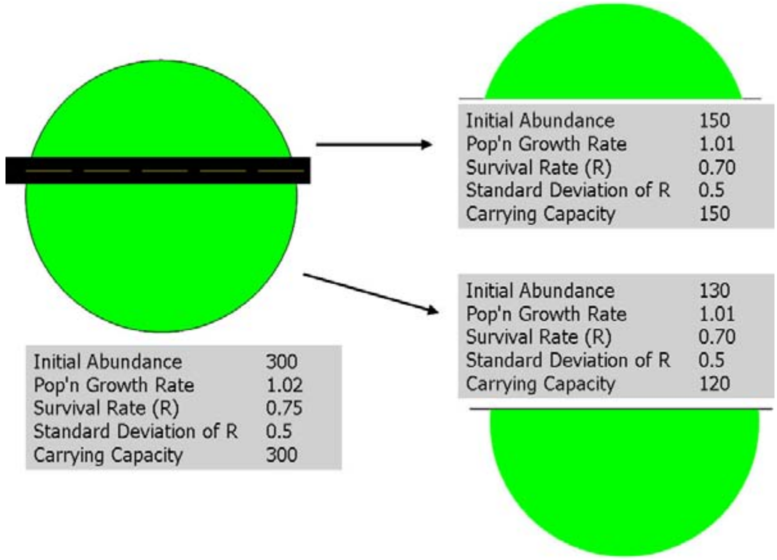
While there may be problems with PVA forecasting, many conclude that, with cautious interpretation of results, it is the best way to proceed (Reed et al. 2003). It can conceptually integrate all information available whether it is based on long-term monitoring studies or expert opinion; models can vary in their complexity as appropriate; and it is explicit about assumptions (Akçakaya 2004; McCarthy et al. 2003). Proponents of PVA argue that alternatives often involve considerable uncertainty that is obscured and that even when data are sparse and predictions less reliable, that the value PVA is in clarifying the problem (Brook et al. 2002). They maintain that alternatives should be evaluated rigorously before acceptance to ensure that their predictions are at least as good as PVA (Lindenmayer et al. 2003). So, while recognizing its limitations use of PVA to integrate what is known about a species to identify critical gaps in knowledge and make testable predictions about which conservation actions have the best chance of reducing a species' predicted risk of extinction is valid even when model predictions are inaccurate (Boyce 1992).

■ *While there may be problems with PVA forecasting, many conclude that, with cautious interpretation of results, it is the best way to proceed (Reed et al. 2003).*

PVA can be used for impact analysis by estimating vital rates for the different scenarios (Figure 4). In this example a proposal is made to build a road through occupied habitat. Assuming that the population parameters are known or can be estimated the text box on the left represents the population before the proposed impact. Then based on available data or expert opinion population sizes, carrying capacity, and vital rates are estimated for the two resulting fragments. If other road alignments are feasible they can be parameterized as well. The model can then be run for the baseline and alternative actions. When habitat loss occurs it can be reflected directly in the carrying capacity and indirectly in the vital rates if the population or habitat patch becomes small enough that survival or

reproduction is impacted. Similarly alternative vital rates are estimated to evaluate alternative management scenarios.

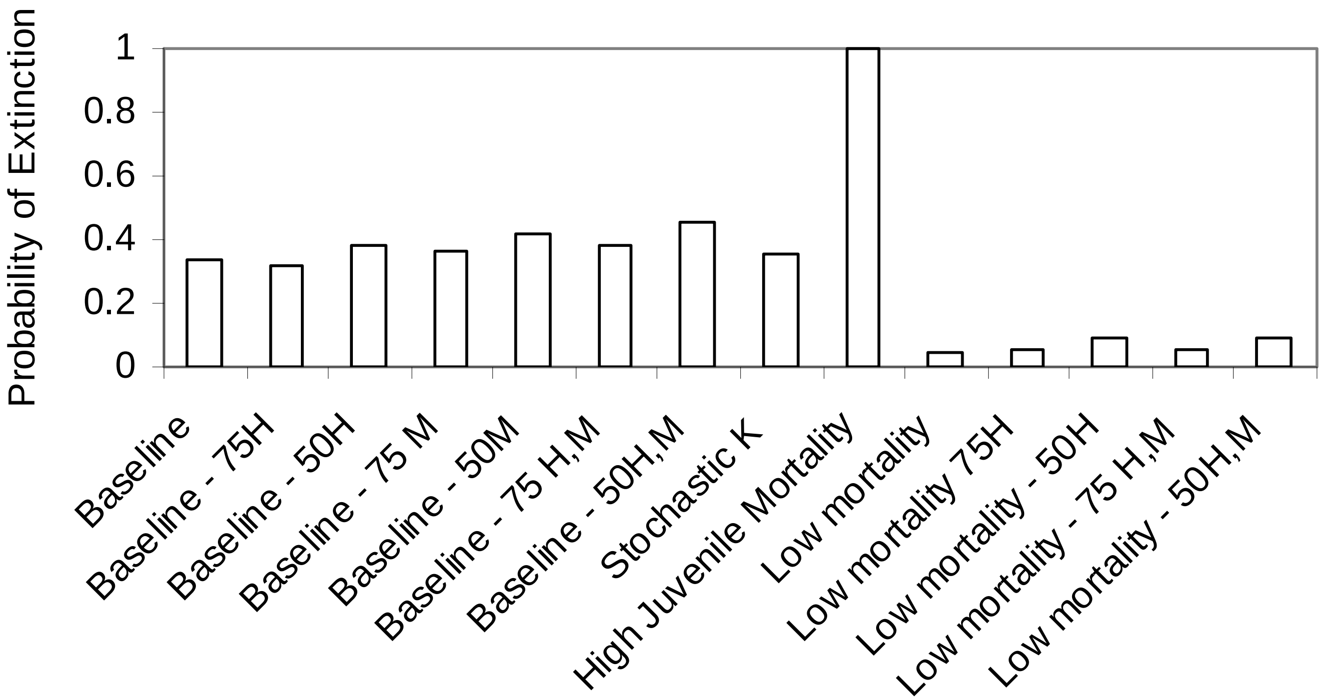
Figure 4. A hypothetical impact analysis where new vital rates are estimated for habitat fragmented by a road.



■ *An example of the application of PVA in DoD land management is the Navy's development and use of a San Clemente Island Sage Sparrow model.*

An example of the application of PVA in DoD land management is the Navy's development and use of a San Clemente Island Sage Sparrow model. The San Clemente Sage Sparrow was listed as a federally threatened subspecies endemic to San Clemente Island in 1977. Habitat for this subspecies is restricted due in part to current and predicted military requirements. Population Viability Analysis was used to evaluate various management strategies to protect the Sage Sparrow population while sustaining requirements for completing the military mission. Habitat for Sage Sparrows exists as high, medium, or low in quality depending on the vegetation (height and cover of boxthorn), elevation and sage sparrow density. The destruction of habitat to ensure the execution of the military mission may be required; therefore a trade-off exists between protecting the threatened species and allowing use of the land. In order to evaluate various management options, models simulating a loss of various amounts of habitat of differing quality and varying juvenile mortality were tested to determine which combination would result in the lowest probability of extinction (Figure 5) of the sparrow. The lowest probability of extinction resulted from a decreased juvenile mortality rate while an increase in the juvenile mortality rate drastically increased the probability of extinction. As long as there was less than a 50% loss of habitat, no matter the quality, the probability of extinction was not severely increased. By clearly defining management options, trade-offs can be highlighted, quantified (in this case by ranking the probability of extinction), and evaluated under a variety of plausible scenarios.

Figure 5. Model simulation results comparing the probability of extinction for San Clemente Sage Sparrows based on various management strategies tested. Baseline represents no disturbance to habitat. The 75H and 50H represent a 25% and 50% loss of high density habitat, 50H and 50M represent a 25 and 50% loss of medium density habitat. Stochastic K represents a scenario where the carrying capacity fluctuates annually. High juvenile mortality is simulated in the model with the highest mortality recorded. Low mortality is simulated as both a juvenile and adult mortality rate 5% lower than the average.



5.0 THE USE OF PVA PREDICTIONS IN CONSERVATION PLANNING:

There are a number of recent examples in the literature where PVA was applied to a conservation planning problem. The most common was the use of PVA to evaluate different management strategies. Akçakaya et al. (2003) used a metapopulation model to evaluate predator control strategies for the California Least Tern. In that case sufficient data had been collected since the development of the California Least Tern model to test its accuracy (Dr. C.T. Collins, pers. comm. January 2006). Evaluation of model accuracy, followed by improvements are needed as part of the adaptive management process and to help maintain stakeholder confidence in the use of models. Other recent examples include the use of PVAs to evaluate alternative management strategies for metapopulations of a snail and a plant. Various harvesting strategies and plantation options were tested to determine the effects on a snail whose habitat is frequently harvested for timber (Regan et al. 2001). Finally, Regan et al. (2003) tested the effects of decreased seed predation and fire management strategies (deterministic and random fire events, mosaic fire management) to a plant.

PVA has also supported the development of criteria for removing an endangered butterfly from the endangered species list (Schultz and

Hammond 2003). In this case the authors modeled a metapopulation using linear regression and calculated the growth rate necessary to have a 95% probability of persistence for 100 years. Their criteria included minimum number of subpopulations, and population growth rate over 10 years with a specified variance.

Problems with PVA that significantly compromise the accuracy of model projections remain. However, there is broad agreement that it is the best process available to make timely decisions with limited data. Sensitivity analyses combined with using a range of parameter values and their variances in modeling exercises can be used to incorporate uncertainty in PVA when accurate estimates are not available. This is probably conceptually close to what experts and managers do when they know their data are uncertain; the advantage is that it is more thorough, transparent and repeatable. The value of spatially explicit metapopulation viability models in conservation planning is demonstrated by its routine use in evaluating conservation management actions and developing recovery criteria for at-risk species.

6.0 MODEL TESTING AND VALIDATION:

- *Both supporters and critics alike agree that tests of absolute predictions are important in PVA, as a means of directing model improvement and informing adaptive management (Coulson et al., 2001; Lindenmayer et al., 2003; McCarthy et al., 2003).*

Both supporters and critics alike agree that tests of absolute predictions are important in PVA, as a means of directing model improvement and informing adaptive management (Coulson et al. 2001; Lindenmayer et al. 2003; McCarthy et al. 2003). Tests however, must not use the same data that was used to parameterize the models (McCarthy and Broome 2000). Model testing results to date demonstrate that while some models can accurately predict population dynamics, accuracy is variable and which models will be more accurate can elude prediction.

Lindenmayer (2003), in an evaluation of spatially explicit metapopulation models found significant variation in accuracy for models of closely related species in the same system. Some models had poor predictive ability even though they dealt with well-studied species. They concluded that model accuracy was limited by complex processes that were poorly understood or unquantified. In contrast Stephens et al. (2002) was able to develop an accurate spatially explicit behavior-based model for the alpine marmot, a species with a complex life-history. This type of model incorporates an optimization criterion to model parameters rather than specified probabilities, such that the way an animal behaves determines the value of the parameter.

7.0 UNCERTAINTY IN PVA

Uncertainty is a major issue in decision making for natural resource management. Methods for robust decision-making under uncertainty should be employed so the manager can gauge the impact of uncertainty on decisions and to highlight where further data collection would have the

most benefit. The data thus targeted for collection can then be used reduce uncertainty in model outputs.

Uncertainty is what causes risk in planning and decision making. In other words, if outcomes were known with certainty there would be no risk that a course of action would produce unanticipated results. Further, understanding the sources of uncertainty and their implications in PVA helps insure the decision process can proceed adaptively.

■ *Uncertainty is what causes risk in planning and decision making.*

Recent reviews of PVA have demonstrated that they are fraught with uncertainty. This stems from several sources, including: 1) measurement error and resultant inaccurate estimation of vital rates, 2) insufficient sample size and 3) assumptions that certain environment drivers (i.e. precipitation, temperature, exotic species introductions, etc.) and its effects on vital rates will remain constant (Coulson et al. 2001; Lindenmayer 2003; Reed et al. 2003).

Measurement error has been identified as a significant risk although Reed et al. (2003) concluded in their analysis of PVAs for 102 vertebrate taxa that variation in vital rates (demographic parameters such as birth and death rates) has a stronger effect than measurement error on extinction predictions (probability of extinction within a specified time frame). Furthermore, estimates of vital rates can be in error if they are calculated from dissimilar populations of the same species. For example, Lindenmayer (2003) found vital rates measured in larger, less fragmented populations to be different than those in smaller more fragmented populations. In general, insufficient sample size and short time series are one of the biggest problems in model parameterization; however, Reed et al. (2003) concluded that insufficient study length causes underestimations of extinction risk rather than simply imprecise estimations, meaning that the results can still be useful in ranking management options.

In addition to measurement error, natural sources of variation introduce uncertainty. Temporal variation in vital rates is not unlikely in natural populations. In order to accept the outcomes of models one must assume that the mean vital rates and their variances are the same for the period from which the model was parameterized and the period for which projections are made. Factors that regulate the population such as predators or prey could change over time as the population increases or decreases, not only influencing the vital rates themselves but which vital rates are most important in regulating the populations (Coulson et al. 2001). The incorporation of climate models and models of exotic species spread to develop better predictions of future environment can help mitigate, but not eliminate this uncertainty.

Typically uncertainty is incorporated in models by using means and variances of vital rates. However, when specific parameter variances are

■ *Sensitivity analysis, also known as perturbation analysis (Caswell 2001), is used to evaluate models to identify variables with the strongest effect on output. It is a tremendously important part of modeling.*

not available it can be incorporated using likely ranges of parameters (Figure 6) and alternative model structures, followed by comparisons of model results to get a range of predictions (Akçakaya 2004; Regan et al. 2003). In addition to incorporating uncertainty into models this is one of a number of techniques that can be used to perform sensitivity analysis. Sensitivity analysis, also known as perturbation analysis (Caswell 2001), is used to evaluate models to identify variables with the strongest effect on output. It is a tremendously important part of modeling. In fact where models are poor because of uncertainty, sensitivity analysis may be the most informative part of modeling in that one can investigate what parameters contribute most to population growth rate. Additionally, sensitivity analysis can reveal parameter thresholds where predictions such as risk of extinction change significantly and thus highlight where it is most important to control uncertainty. This information is useful for prioritizing conservation actions as well as for allocating effort to improve estimates of these values. The variables addressed can be for example vital rates, age or stage structure, density dependence or dispersal (Mills and Lindberg 2002).

Figure 6. A hypothetical sensitivity analysis where the juvenile survival rate was perturbed plus or minus 10% to evaluate the effect on model output.

Parameter	Extinction Risk
Juvenile survival (S_J)	45%
(S_J) + 10%	24%
(S_J) – 10%	55%

Calculation of confidence intervals (Ludwig 1999) is an explicit way to express the uncertainty of model results. When the uncertainty is high it may not be possible to differentiate between estimates of population viability and it follows that without such estimates of precision, specific estimates of viability should not be used (Fieberg and Ellner 2001). In those cases, model output should be expressed as relative measures of viability for alternative impact or management scenarios and these relative measures used to inform decisions.

8.0 SINGLE SPECIES MODELS

■ *PVAs have progressed from deterministic evaluations of single populations (single species models) to stochastic models and, more recently, spatially explicit metapopulation models and individual-based models (Beissinger 2002).*

Even though our purpose here is to address multi-species modeling techniques for DoD INRMP applications, a review of single species models is important to lay the methodological foundation for discussing of multi-species modeling. PVAs have progressed from deterministic evaluations of single populations (single species models) to stochastic models and, more recently, spatially explicit metapopulation models and individual-based models (Beissinger 2002). In deterministic models, a

single population trajectory is calculated from mean vital rates, such as birth rate and death rate.

However, using averages and a single trajectory does not usually represent the behavior of natural populations very well. Stochastic models incorporating variance are necessary to describe population dynamics because of high variances in vital rates (i.e. population growth, birth, death and survival) of wild populations and their interaction with non-linear population dynamical functions (Boyce 1992; Chesson 2000). Non-linearities are common in natural systems, for example in perennial plants, year to year survival buffers population growth rate from the recruitment rate so when there is no recruitment the population doesn't necessarily go extinct. On a simplistic level non-linearities can mean that the good years outweigh the bad because the population gains much more in the good years than it loses in bad years. With stochastic models, multiple trajectories are produced (Figure 7) each one based on calculations using birth and death rates drawn from a probability distribution defined by the mean and standard deviation of the given vital rate. It is typical for a trajectory summary to be based on 100s to 1000s of trajectories. The results are given as the average expected population size and its standard deviation over time (Figure 8). Spatially explicit metapopulation models calculate a species' population trajectories based on calculations for individual populations which include factors for migration between patches and degree of correlation between patches of environmental factors which influence annual population fluctuations. Individual-based models follow the fate of individuals through time.

Figure 7. Multiple trajectories produced by a stochastic population model.

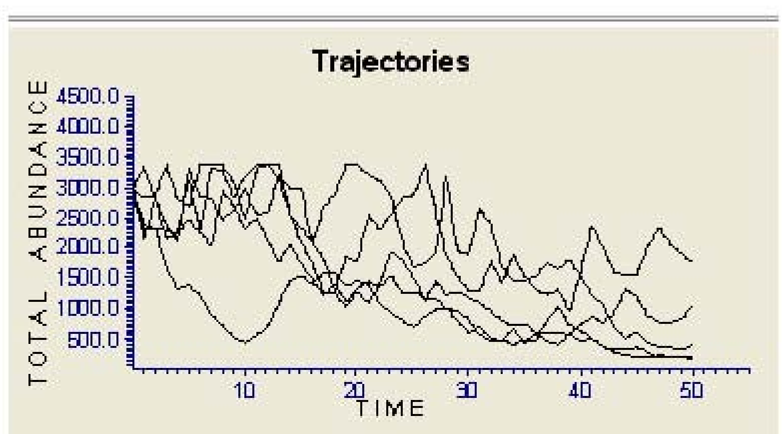
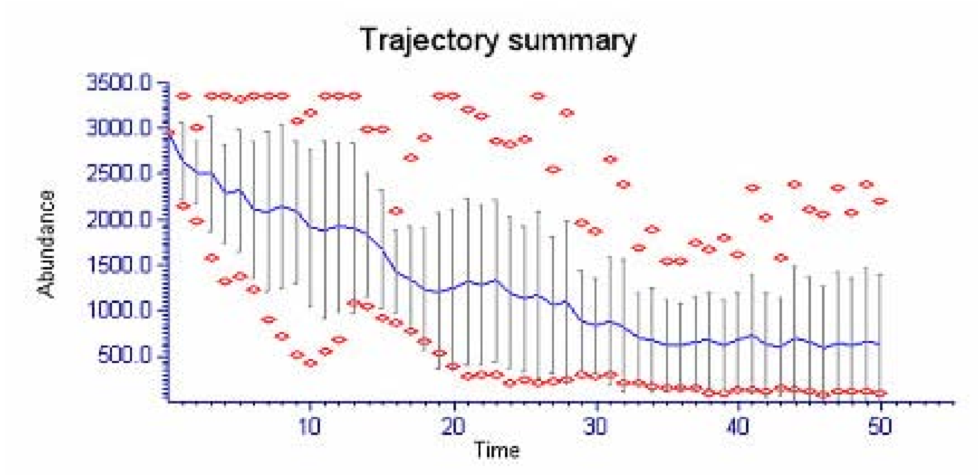


Figure 8. Trajectory summary showing the average abundance at each time step and 1 standard deviation.



- *Different management or impact scenarios are evaluated by altering vital rates in the models and comparing predictions.*

Different management or impact scenarios are evaluated by altering vital rates in the models and comparing predictions. Estimates are made for how these scenarios will change vital rates or carrying capacity and these new values are entered into the model. For example, Figure 9 shows some of the parameters that are used in the RAMAS metapopulation model (Akçakaya et al. 1999). Figure 9 represents a hypothetical impact which fragmented a single population into two. The parameters that were changed to represent the impacts included initial abundance, population growth rate, survival rate and carrying capacity. In this case the impact was anticipated to result in: the loss of individuals (initial abundance), the loss of habitat (carrying capacity), an increase in mortality (decrease in survival rate) leading to a reduction in the population growth rate. These predicted estimates, as with the original model parameters, can be based on information from a variety of sources from studies to expert opinion. Uncertainties can be incorporated by conducting sensitivity analyses. For example if the available information indicates that the survival rate will decrease between 0.05 and 0.15 the model can be run separately using both figures and the effect on risk of extinction can be quantified. Uncertainty can also be incorporated by adjusting the predicted standard deviation of the population growth rate.

Similar to quantifying an impact analysis, management options proposed to promote population persistence can be evaluated in this manner. The optimal management strategy will be that which results in the population most likely to persist.

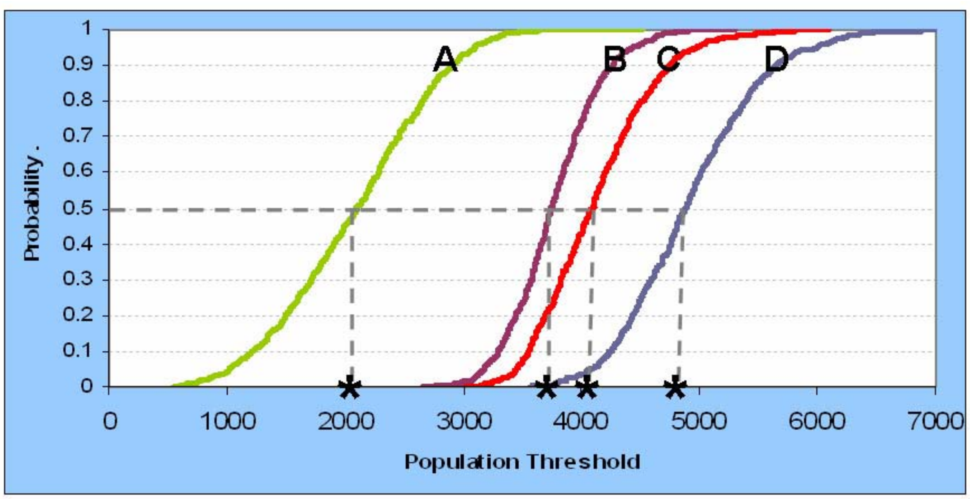
Once the parameters have been estimated for the different scenarios, each model is rerun. Average population trajectories are calculated for each of the management options or impact scenarios. From these trajectories for

each alternative, risk curves can be developed by calculating the cumulative probability that the population will fall to or below a certain size based on the individual trajectories (Akçakaya 1999). Alternative management scenarios are compared by considering the probability that a population will fall below a given threshold over the period modeled. Since we are most concerned about the worst case scenario for at-risk species, risk curves are generated to conservatively compare management options. Figure 10 shows hypothetical risk curves for four management scenarios. The curves allow one to evaluate the alternatives with respect to each other. For example, while management scenario A has a 50% chance of falling below 2000 individuals, that threshold is higher for management scenario B (3,800), and C (4,100) and management scenario D achieves the highest population threshold of 4,900 individuals.

Figure 9. Parameter Input

Population	Initial Population	Fragment 1	Fragment 2
Initial Abundance	300	***150***	***130***
Population Growth rate (R):	1.02	***1.01***	***1.01***
Survival rate (s):	0.75	***0.7***	***0.7***
Standard deviation of R:	0.5	0.5	0.5
Carrying Capacity (K):	300	***150***	***120***

Figure 10. Risk Curves. The dotted lines show the population threshold for each management scenario for a probability of 0.5.



Metapopulation PVAs evolved to address spatially oriented management questions for populations with patch structure and are a key development in conservation biology. Natural populations have always had a certain degree of fragmentation (Schwartz 1999). However, fragmentation caused by human activities has proceeded to the extent that Reed et al. (2003) concluded, based on their MPV estimates for 102 vertebrate taxa, that sufficient contiguous habitat is not available for most species and that

■ *Metapopulation PVAs evolved to address spatially oriented management questions for populations with patch structure and are a key development in conservation biology.*

management of habitat networks is required. Because they consider the contribution of each patch to the persistence of the species, spatially explicit meta-population models can be used to evaluate management of and impacts to these populations. These models evaluate extinction risks for species dependent on fragmented habitats taking into consideration dispersal, environmental correlation among patches, variations in population growth rates and differences in carrying capacity between patches. Figure 11 shows an example of a metapopulation.

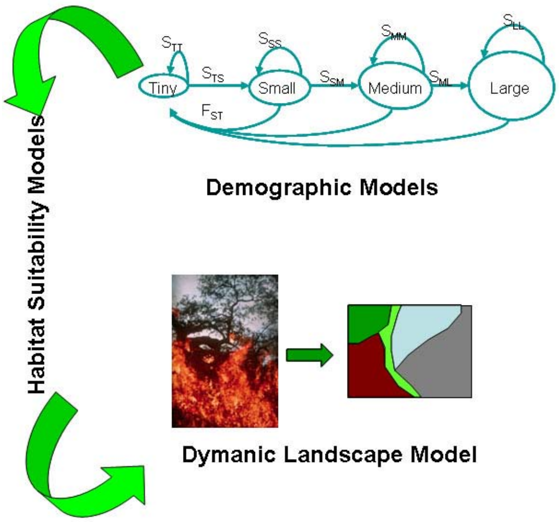
Figure 11. Metapopulation Map

↔ Arrows show dispersal between patches.

■ *Recent efforts link models of spatial and temporal habitat dynamics at the landscape scale and metapopulation models that predict occupancy at the patch level.*

At a landscape scale, habitat patch dynamics have a strong influence on population persistence and the predicted risk of extinction and therefore, should be included in PVAs as well (Shaffer 1990). Fire, for example, a strong driver of habitat dynamics, is highly variable over time and space. In order to model populations more accurately, changes in habitat suitability must be reflected in changes in carrying capacity for given patches. When a patch becomes unsuitable, the carrying capacity changes to zero. If a patch becomes suitable because of, for example a disturbance, the carrying capacity will increase to a positive number but the population size will remain at zero until successful dispersal occurs. Recent efforts link models of spatial and temporal habitat dynamics at the landscape scale and metapopulation models that predict occupancy at the patch level (Figure 12) (Akçakaya et al. 2004).

Figure 12. Habitat suitability models are used to link dynamic landscape models to demographic models so that predictions of risk include habitat as well as demographic dynamics.



9.0 MULTI-SPECIES MODELING AND MANAGEMENT

9.1 Multi-species Management

The conservation of biological diversity is one of the primary conservation goals not only for government and non-government organizations devoted to conservation but also the U.S. Military (Williams 2000; Leslie et al. 1996). Definitions of biodiversity encompass biotic variation from genes to landscapes and ecological and evolutionary processes (Noss and Cooperrider 1994). Conserving biodiversity has shifted away from a paradigm which focuses on counting species and the perception of ecosystems as static and predictable. The current view is of a dynamic, complex system with multiple levels of organization (Poiani 2000). Realization of the complexity and dynamic nature of ecological systems led to the concept of ecosystem management, wherein success is assured by conserving and managing the ecosystem as a whole (Christensen and Franklin 1996). This is problematic however, because as a goal, ecosystem management is too vague to set meaningful objectives by which progress can be measured.

- *Conserving biodiversity has shifted away from a paradigm which focuses on counting species and the perception of ecosystems as static and predictable. The current view is of a dynamic, complex system with multiple levels of organization (Poiani 2000).*

Conventionally, natural resources management has focused on individual species for two reasons: 1) the complexity of considering multiple species simultaneously until recently outstripped theory and available tools and 2) the imperative of intensively managing at-risk species to avoid extinction focused resources on individual species. We will focus on the first here. Unfortunately so many species have become endangered that the density of federally listed species on military lands has put the military in the position of managing multiple listed species, in some cases with

- *Unfortunately so many species have become endangered that the density of federally listed species on military lands has put the military in the position of managing multiple listed species, in some cases with unavoidable trade-offs between them due to conflicting habitat or management requirements (U.S. Navy 2007).*

unavoidable trade-offs between them due to conflicting habitat or management requirements (U.S. Navy 2007). Furthermore, when one considers the importance of managing rare species to forestall future listings in addition to already listed species, multiple species management has become the imperative for the military. This perspective, focusing on at-risk species, is narrow compared to the business of ecosystem management and developing and implementing INRMPs, but it is where we need to start.

- *Switching from single to multi-species management magnifies the problems of single-species modeling (Nicholson and Possingham 2007).*

Switching from single to multi-species management magnifies the problems of single-species modeling (Nicholson and Possingham 2007). Data limitations for example, mean that not all species can be modeled and included in the development of management plans. Because of this, surrogates are necessary but their selection is often biased, if for no other reason than the fact that more data exist for certain types of species such as those that are charismatic or protected. This can leave plans with incomplete coverage of ecosystem components and processes.

The use of species guilds was an early multispecies approach to conservation and management (Poiani 2000). The species guild concept, which focuses on a group of related species and the umbrella species concept (Andelman and Fagan 2000; Simberloff 1998) which focuses species requiring large areas that incorporate the ranges of other species, were a substantial improvement over traditional single species approaches but still had their limitations. Sites conserved and managed for the needs of single or small group of species may fail to conserve critical components (habitat elements, linkages and processes) that other portions of the biotic community are dependent on. In trying to assess whether ecosystems have the ability to maintain species over the long-term, interest has developed in indicator species (Poiani 2000) as a way of meeting the needs of many species without studying them individually (Lambeck 1997). However, because species requirements do not always nest within each other indicator or focal species can be problematic (Lindenmayer 2002).

Lambeck (1997) addresses the problem of the unproven assumption that conservation of one species will protect other species in the same habitat by proposing the following selection rules for the “focal” species. Each species included in the set will represent groups of species with similar habitat and process requirements and threatened by similar factors. The entire set of species should represent the different aspects of the landscape. Through this process the species with the most limiting requirements will be selected and appropriate management and/or restoration activities can be implemented. Theoretically, the species chosen through this method will define different attributes of the landscape necessary to meet the needs of its biota. Once a management plan is developed based on the

focal species selected, an adaptive management program must be implemented wherein both focal species and the others should be evaluated to see if the assumption that other species will be protected holds.

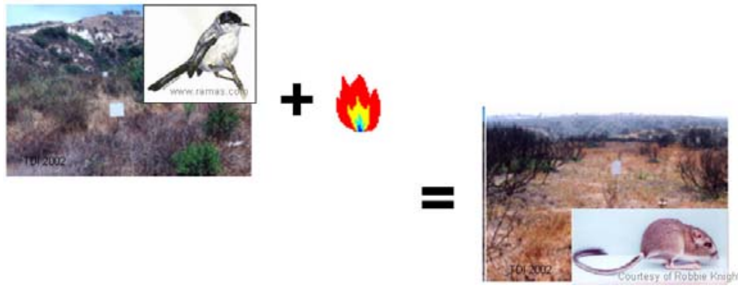
Fox et al. (2004) used a similar approach to Lambeck (1997) to select study species. In evaluating the effect of forest management they chose species that were anticipated to be at risk from logging and which had varying life histories and ecological characteristics. They anticipated that this focus on species with a range of habitat requirements would overcome drawbacks of single species approaches. Methods to select the best species not withstanding, a problem noted by many authors is that the species chosen are biased by the available information. While some choices must be made, this issue is less problematic for DoD at this time, where because of the imperative of ESA compliance while simultaneously achieving the sometimes conflicting military mission, the focus has been narrowed to listed or likely to be listed species.

9.2 Trade-offs and Goal Conflicts

Implicit in multi-species management are trade-offs. Whereas trade-offs in single species management occur between conservation of the target species, social values (such as competing land uses) and economics, with multiple species trade-offs also occur among the targets of conservation. Trade-offs can occur between the objects of conservation (Figure 13) such as deciding where limited conservation funds are spent, or between conservation and other objectives such as military training or development. In some contexts, species conservation may be the primary management goal. In other contexts management may also be focused on harvesting natural resources or other land uses such as military training. Regardless of whether the goals are limited to multiple species conservation or multiple species conservation plus other resource use, trade-offs between management objectives must be considered. Trade-offs emerge when conflicts exist between management objectives. They often happen when there are multiple stakeholders with different interests, but also when a single decision maker has conflicting objectives. For example, military managers would like to train without restrictions but at the same time they want to prevent soil erosion that reduces the utility of training lands either by physically reducing access in the form of gully erosion or by changing plant communities which might be important in training scenarios.

- *Implicit in multi-species management are trade-offs.*
- *Trade-offs can occur between the objects of conservation such as deciding where limited conservation funds are spent, or between conservation and other objectives such as military training or development.*

Figure 13. Trade-offs in habitat management between the California Gnatcatcher (CAGN) and the Stephen's kangaroo rat (SKR). In some areas of coastal sage scrub habitat short fire return intervals make the habitat suitable for the SKR while longer fire return intervals exclude the SKR and allow development of habitat suitable for the CAGN.



Conflicts between conservation objectives can include both trade-offs among species that utilize the same landscape in different seral stages and trade-offs between species that are inexpensive to conserve and those which are more expensive to conserve. For example, one of the trade-offs often made within DoD is controlling native predators or altering native habitats to support endangered species. The Sikes Act in its language governing INRMPs acknowledges trade-offs by saying that implementation of the plans shall result in no net loss of military training opportunities. In that case, the word "net" implies that the decision process can include trade-offs between military training, and other objectives such as conservation and economics.

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9.3 Multi-species Modeling

Once a subset of species has been identified as the focus of management, methods must be employed that allow their simultaneous evaluation so that trade-offs can be analyzed to support decision making. One of the key challenges in developing multi-species models is to define objectives clearly and explicitly. Objectives provide the context that allows trade-offs to be clarified. Maximizing biodiversity is too vague and needs to be translated into specific objectives. On the other hand, an objective such as a specified risk of extinction allows a clear comparison of alternatives. In addition to the importance of specificity, how objectives are expressed will influence output. Nicholson and Possingham (2007) recommend using the relative risk of extinction to define objectives for multiple species. They use the objective functions to compare the risks within a group of species and management options to accomplish various objectives such as maximizing the number of species expected to persist or minimizing the probability of extinction for any given species. Below we review two theoretic and two applied approaches that utilize relative risk of extinction to direct conservation planning. The applied approaches develop individual PVAs for each species and the relative results of those PVAs are compared (Fox et al. 2004) or combined (Akackaya 2000). Both theoretical examples involve more complex functions.

Fox et al. (2004), in a study on the effects of a conservation plan that increases logging while at the same time provides for conservation, constructed PVAs for 11 rare and endangered species. The premise was that increased impacts caused by increasing timber production in certain areas could be offset by species benefits in others. They evaluated the contribution of spatial configuration versus quantity of habitat to species persistence and hypothesized that they would differ in importance to different species. They used PVAs constructed for each species and based their conclusions on the relative change in risk of extinction for each species predicted from alternative management scenarios.

Using objective functions that directly incorporate the extinction risk of multiple species, Nicholson and Possingham (2007) took the results of PVA and created mathematical formulations of conservation objectives. These included for example, conservation based on umbrella species (Andelman and Fagan 2000; Simberloff 1998), or minimizing the chance of extinctions in a group of species. The function for minimizing the chance of extinctions is set to “minimize the probability of one or more extinctions.” The results are used to rank conservation plans and to solve optimization problems. They showed that alternative functions can lead to different results. For example, the choice between the alternative functions 1) minimize the chance of one or more extinction or 2) minimize the chance that all species go extinct resulted in a reversal of the ranking of the best and worst scenarios.

One of the problems Nicholson and Possingham (2007) encountered was that correlations in extinction risk are poorly known and they need to be explicitly treated when joint probabilities (multiplicative) are used. As a result they recommended combining the threshold approach (their umbrella species approach) and the use of an additive function such as minimizing of the expected number of extinctions rather than the joint probability. Finally they cautioned against taking the final scores at face value and emphasized the need to look at the impacts of the various scenarios on individual species in making decisions.

Akçakaya (2000) uses PVA with weighted habitat suitability maps to develop multi-species plans. The value of each habitat patch to the viability of each species is calculated using a model with all patches and a model with all patches except the one under consideration. The values are weighted by the threat status of a species and are combined for each parcel. The resultant values therefore are higher for areas with more species and areas with more threatened species.

Witting (2000) proposed a method to provide optimal protection from extinction within a set of interacting species. They note that optimal protection might require trade-offs such as an increase in the extinction

risks, or even an eradication of species that either compete with or prey on other species. They analyzed the effects that single species extinctions have on the extinction dynamics of the complete ecosystem. In their method, an ecological value is given to a species by the degree to which the extinction of a species causes or prevents the subsequent extinction of other species.

10.0 INCORPORATING CLIMATE CHANGE

- *Addressing climate change poses a new challenge for natural resources managers who will need, in addition to understanding ecosystems as they function now and in the past, to anticipate changes in ecosystem structure and function (GAO 2007).*

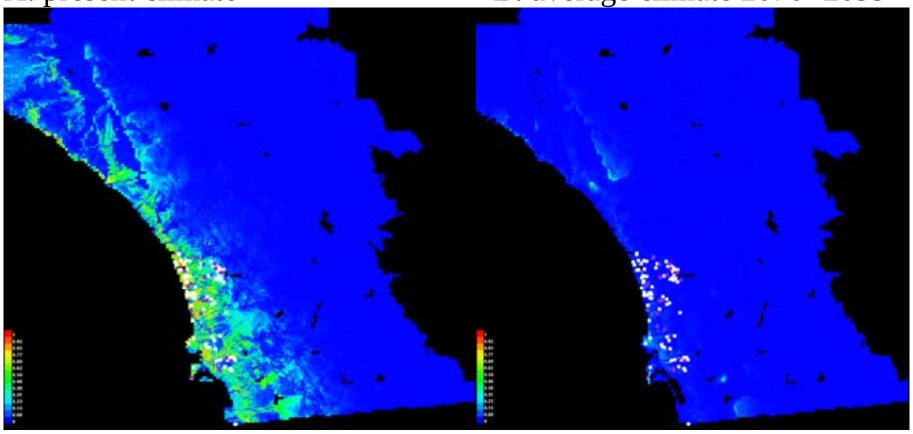
- *Because of the uncertainty it is very important to remember that these models should be used to support rather than guide decisions (Dormann 2007, and Guisan and Thuiller 2005).*

The evidence for human caused climate change is extensive and has generated consensus in the scientific community (GAO 2007; Gitay et al. 2002; and Oreskes 2004). Addressing climate change poses a new challenge for natural resources managers who will need, in addition to understanding ecosystems as they function now and in the past, to anticipate changes in ecosystem structure and function (GAO 2007). Models are the only way to project these future changes, however the use of models to explore the potential implications of climate change is rife with uncertainty. To begin with the climate models themselves are uncertain and vary in their predictions (Cayan et al. 2008). In addition the use of current data to model future changes relies on questionable assumptions including constancy of limiting factors and interspecific competition, and that the species will not evolve in response to climate change. Because of the uncertainty it is very important to remember that these models should be used to support rather than guide decisions (Dormann 2007, and Guisan and Thuiller 2005). The problems and limitations not withstanding, it is important to move forward, improving the ability to project future changes; without this managers will be limited to reacting to changes as they occur and constrained in their ability to plan around future changes. The adaptive process of developing, validating and improving models is the best way forward to improve forecasts needed for management.

One of the main methods used to date to assess the impact of climate change on biodiversity are projections of species distributions or habitat suitability models (HSM's) where current distributions of species are correlated with climate and sometimes other habitat predictor variables such as soils (Dormann 2007; Keith et al. in press; and Thomas et al. 2004). The statistical models developed are then used with future climate scenarios to project range shifts (Figure 14). One of the problems with these projections is for example, that for a long lived woody species like *Ceanothus verrucosus* the decline in suitability may only affect certain life stages such as seedlings so the adults might be expected to persist in their former range even if the habitat becomes unsuitable for seedling establishment. Dormann (2007) and Guisan and Thuiller (2005) provide a good discussion of the limitations of these models.

Figure 14. Habitat suitability model for *C. verrucosus* under present conditions and future conditions predicted using the National Oceanic and Atmospheric Administration's model GFDL CM2.1. Relative habitat suitability is indicated by a color ramp from red (highly suitable) to dark blue (unsuitable).

A. present climate B. average climate 2070 -2099



Recently, HSMs have been combined with population models to improve predictions of extinction risk under climate change (Keith et al. in press). For each species they developed an HSM and stochastic population model. Habitat suitability maps for specific climate change scenarios were then derived using the HSMs. The carrying capacity of the projected suitable habitat was determined two ways, either with a combination of habitat area and relative habitat suitability or just habitat area. They used linear interpolation to derive maps for each time step in the population model and population dynamics were simulated using the combination of the population model and projected habitat. These methods require a good understanding of both the habitat requirements and life-history of the species' modeled, but for species with sufficient information they offer a promising way to improve projections under climate change by integrating projected habitat changes with population dynamics.

11.0 DECISION THEORY

Integrated natural resources management planning is a classic application of decision theory, though most people involved with these plans do not explicitly think about it as such. Decision theory includes principles and analytical techniques that facilitate selection among alternatives based on their consequences. Key aspects include values of the decision maker, risk and uncertainty. Application of decision tools clarifies the logic and facts that support decisions. Resnik (1987) describes it as an integration of utility theory (how we value things), game theory (strategic decisions) and social choice theory (voting systems). Not all decisions call for a methodical application of decision theory. In some cases the choices and outcomes are so limited or clear cut that it is rational to make a direct decision, or immediate need or danger might not allow time to run through a decision analysis such as the case with a natural disaster like fire or

- *Decision theory includes principles and analytical techniques that facilitate selection among alternatives based on their consequences.*
- *Application of decision tools clarifies the logic and facts that support decisions.*

flood. We will address the cases where sufficient time is available to complete a decision analysis and the complexity and the risks associated with the choices warrant it.

- *Decision makers do not influence the objective truth of a situation or the probability of an outcome. Their challenge is to maximize the benefits received from positive outcomes, minimize the adverse effects of negative outcomes and avoid unacceptable outcomes.*

To start with it is important to acknowledge that decision makers do not influence the objective truth of a situation or the probability of an outcome. Their challenge is to maximize the benefits received from positive outcomes, minimize the adverse effects of negative outcomes and avoid unacceptable outcomes. They can do this best by having all of the available facts arrayed as clearly as possible in a conceptual framework or model with logical connections explicitly specified. Decision theory itself is conceptually simple; select the alternative with the highest value or utility. The difficulty is in establishing the decision maker's objectives and preferences, considering multiple objectives that cannot always be resolved to a single common denominator, developing alternatives and determining consequences of the alternatives.

Decision analysis is typically used to classify items (such as species in threat categories), select from a group (such as a set of reserves from potential habitat) and to rank actions such as management options. Within the INRMP context, selection and ranking options are the most useful. When comparing actions it is important to consider not only the effects, but the magnitude and timing of the effects. Timing can be especially important when considering economic/environmental trade-offs such as short-term economic benefits with long-term environmental costs. Often the long-term predictions, whether they are environmental or economic, have much greater uncertainty than short-term predictions. Differences in time preferences often result in differences in rank order of management options.

- *Realistic problems that require decisions typically include some level of subjectivity (Drechsler and Burgman 2004). While subjectivity is often viewed negatively, all values are inherently subjective.*

Decision analysis takes the broader view of coming to a decision based not only on the facts but the objectives and values of the decision maker(s) (Drechsler and Burgman 2004). Multi-criteria decision analysis considers the positive and negative aspects of multiple points of view. One of the challenges of multi-criteria analysis is amalgamating the sometimes conflicting and subjective preferences of individual stakeholders and decision makers (Figueira et.al. 2005). Realistic problems that require decisions typically include some level of subjectivity (Drechsler and Burgman 2004). While subjectivity is often viewed negatively, all values are inherently subjective. Because of this, subjectivity is often introduced in decisions through the preferences and values of stakeholders and decision makers. Advocates of military readiness, for example, may believe that when military readiness and endangered species protection are in conflict that military readiness should prevail because it has higher value. Conservation advocates could be expected to hold the opposite view. Regardless, while subjectivity should be explicitly addressed in the

decision framework, it is typically viewed negatively and this is not always done.

There can be merit in running through a decision framework using more than one set of values to see exactly where and how often the differences in values result in a different decision (Joubert et al. 1997). Solutions then can focus on the situations where the decisions are affected and not necessarily require wholesale and possibly unrealistic value shifts in stakeholders. Decision analysis can be particularly helpful identifying compromises in situations with multiple stakeholders holding conflicting values. This is known as decision support and relies on clearly identifying stakeholder preferences and options that are consistent with those preferences (Drechsler and Burgman 2004).

The context in which decisions are made can range from simple to complex. Complicating matters in addition to value conflicts is the uncertainty associated with our understanding of ecosystem condition and its response to environmental processes (stressors) and management (Burgman 2005; Drechsler 2004) A third factor that complicates decisions is that they are not all static. Dynamics in decision making occur because values placed on certain outcomes by stakeholders can change over time and the states of the system can change. Adaptive management can help, both when decisions change through time due to system changes as well as when the answer exists but there is so much uncertainty about the state and dynamics of the system that the answer is not clear.

One of the ways that uncertainty is dealt with in decision making is by making conservative decisions. Conservative decisions are called for in the Endangered Species Consultation handbook (USFWS and NMFS 1998). A conservative decision is one that protects against unacceptable or negative outcomes. Conservative decisions address uncertainty by using parameter values that might not be the most likely to occur but are within the realm of possibility and are associated with the worst outcome. Worst case analysis is the most conservative analysis and is applied when there are severe uncertainties. In conservation applications, conservatism is usually justified in terms of avoiding unacceptable outcomes. While conservative decisions can help avoid some negative outcomes there are negative consequences to conservatism as well. These include lost opportunities because of the potentially high economic or social costs associated with the chosen action. The degree of conservatism in a decision typically depends on attitudes to risk, the extent of uncertainty in the problem and the trade-off between costs and benefits. While conservative decisions may be warranted on a case by case basis, the problem with a series of conservative decisions is that they are unlikely to optimize conservation programs because of the opportunity cost to beneficial actions that may otherwise have been taken (Burgman 2005; Halpern 2006).

- *A conservative decision is one that protects against unacceptable or negative outcomes. Conservative decisions address uncertainty by using parameter values that might not be the most likely to occur but are within the realm of possibility and are associated with the worst outcome.*

- *While conservative decisions may be warranted on a case by case basis, the problem with a series of conservative decisions is that they are unlikely to optimize conservation programs because of the opportunity cost to beneficial actions that may otherwise have been taken (Burgman 2005; Halpern 2006).*

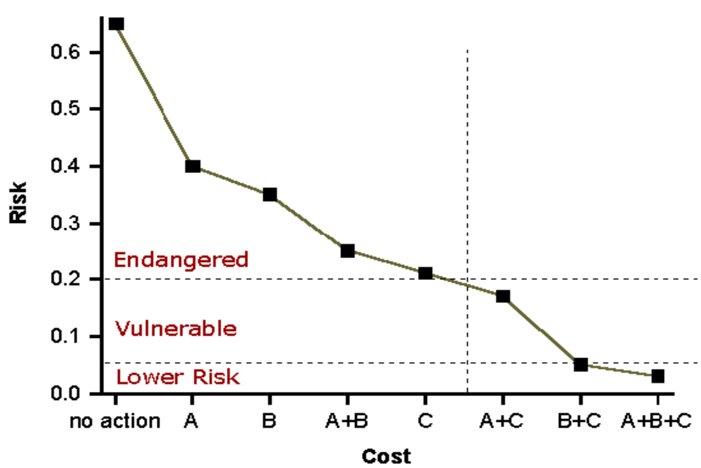
12.0 MULTI-CRITERIA DECISION METHODS

As discussed earlier, trade-offs are central to multi-species management. Multi-criteria decision analysis (MCDA) provides a systematic analysis of trade-offs when selecting among alternative management actions. MCDA is a collection of methodologies that sets out to: 1) portray decision maker preferences over multiple criteria; 2) comprehensively compare alternatives with respect to preference criteria; and 3) present the alternatives and their performance across the criteria for the decision maker. With respect to the third point it is important to note that MCDA does not necessarily "solve" a particular problem; rather its strength is in clarifying performance of the alternatives across the objectives and trade-offs among objectives so that the decision maker can make an informed decision. Decision aiding consists of evaluating the stakeholder's values, goals and objectives; enumerating management options their consequences and risks; structuring the decision process to identify and array values, goals and objectives; fostering cooperation through "mutual understanding" and developing a framework that provides for constructive debate; and clarifying recommendations with results taken from models. This process when done skillfully provides legitimacy (Roy 2005).

- *One of the benefits of considering multiple objectives is that the different objectives do not have to be reduced to a single common denominator such as cost. As with the value of the persistence of a species, it can be difficult to put a monetary value on military readiness.*

One of the benefits of considering multiple objectives is that the different objectives do not have to be reduced to a single common denominator such as cost. As with the value of the persistence of a species, it can be difficult to put a monetary value on military readiness. In fact, the need to make decisions in situations with multiple criteria that are not easily normalized is the rule rather than the exception in natural resources management. Akçakaya (2000) in his work on California Gnatcatcher presents ecological risks and economic costs graphically on separate axes (Figure 15). This depicts the trade-off but doesn't attempt to put an economic value on species persistence. This is left for the decision maker(s).

Figure 15. Risk of extinction (y-axis) vs. costs of conservation alternative (x-axis) (from Akçakaya 2000).



Basing a decision on a single criterion is almost unheard of in this arena where conservation and social agendas often conflict. In some cases criteria can be converted to a common scale, such as currency, but this can give a false sense of objectivity and is often unrealistic. For example, the simplifying assumptions that would have to be made to convert the value of a prepared military and the value of persistence of an endangered species to a common scale such as money would make the resulting values highly unrealistic and suspect due to the difficulty of quantifying such values and conflicting legal mandates. Aggregated indices can cause problems because the metric may be better for one and worse for the other, with the reasons may be totally obscured.

When trade-offs are present there is not a unique solution that returns the highest value for all objectives. In this situation decision makers look for the solutions that cannot be improved on with respect to any one objective without degrading one or more of the remaining objectives. These solutions are known as pareto optimal solutions from the game theory literature (Bierman and Fernandez 1998) and have been used in natural resources policy analyses (Enriquez-Andrade and Vaca-Rodriguez 2004). Figure 16 shows a hypothetical set of solutions for a simplified military situation. This set of pareto optimal solutions shows the range of trade-offs that must be considered and are the ones that decision makers must consider to select a course of action. The selection of a single solution from the set involves subjective determination of the value of the objectives (Enriquez-Andrade and Vaca-Rodriguez 2004). Note that alternative III is not pareto optimal because alternative II is superior in all objectives except cost where it is equivalent. Alternative III therefore, would never be selected by a rational decision maker.

■ *When trade-offs are present there is not a unique solution that returns the highest value for all objectives.*

Figure 16. Pareto optimal solutions for alternatives with multiple objectives. Yellow highlights indicate the best performing alternative for a given criteria. Blue highlighting indicates the set of pareto optimal solutions. Rank order should be used when a reliable estimate of accuracy cannot be made (Fieberg and Ellner 2001).

Alternatives	Cost	# days w/ training restrictions	Probability of Survival species a (rank order)	Probability of Survival species b (rank order)
I	1,000,000	15	2	1
II	500,000	30	3	3
III	500,000	60	1	2
IV	50,000	90	4	1

In order to decide among the pareto optimal alternatives in Figure 16 multiple factors must be considered. Can the military training be achieved elsewhere and what is the essential number of training days without restrictions at this facility? How much money is available to implement

this management plan? The selection of a solution in the above example will reflect subjectivity in how the decision-maker(s) value the objectives.

Another common way to optimize decisions is to develop a framework where one objective like cost is held constant and the other objectives are evaluated in that context. Nicholson and Possingham (2007) used the maximum coverage approach (best conservation outcome for a given budget) to reveal trade-offs between species, conservation and socio-economic outcomes. In addition, thresholds can be used to ensure that there are no unacceptable outcomes (costs too high, military training too restricted, endangered species risk). Typically in conservation applications we would list management options and optimize; however in our application we must balance or trade off between economics, military training and conservation.

- *Multi-criteria decision analysis formalizes the selection of compromise solutions, and is designed to clearly display trade-offs in the management options under consideration (Joubert 1997; Munda in Figueira et al. 2005).*

In summary, multi-criteria decision analysis formalizes the selection of compromise solutions, and is designed to clearly display trade-offs in the management options under consideration (Joubert 1997; Munda in Figueira et al. 2005). Multi-criteria analyses, which include multi-attribute value theory, hierarchy of goals and outranking, can be used to analyze goal conflicts in several ways. Multi-attribute value theory considers the utility of each objective and supports compromises through trade-offs between objectives. Hierarchy of goals is a threshold approach comparing actions based on the performance of the most important objective. Outranking allows limited trade-offs between objectives and uses pair wise comparisons of all actions for each objective (Drechsler 2004; Burgman 2005).

The benefits of MCDM include facilitation of public participation, avoidance of the need to convert disparate values to a common scale, and clarification of decision problem by maintaining separate criteria in an accessible format.

12.1 Value Conflicts

- *Environmental decisions, particularly those involving endangered species, are often made under competing or conflicting social values such as conservation versus property rights or military readiness.*
- *Identifying suites of actions that meet the goals of the military, regulatory agencies and other stakeholders is necessary for progress.*

Environmental decisions, particularly those involving endangered species, are often made under competing or conflicting social values such as conservation versus property rights or military readiness. The military context is no different where there typically are multiple stakeholders with conflicting values. Playing a role at a minimum are the military with their primary mission of defense readiness; regulatory agencies mandated to protect listed species; and increasingly, non-governmental advocates for conservation. Identifying suites of actions that meet the goals of the military, regulatory agencies and other stakeholders is necessary for progress. Identifying innovative solutions under these circumstances is critical. Decision support tools that establish transparent frameworks help stakeholders clarify key issues and move beyond established positions. When stakeholders better understand each other's needs and the reasoning

behind them they are better able to identify solutions that maximize benefits for all parties (Burgman 2005).

One problem faced by decision makers is that certain value conflicts play out as goal conflicts such as the maintenance of military readiness and conservation of endangered species. This happens because the choice of goals is based on values. For example an active deer hunting program may require prescribed fire with fire return interval that is not optimal for management of a certain endangered species. The hunting advocate may maintain that a specific fire interval is better for the environment whereas the endangered species manager may advocate for a different management scheme. When the goals and objectives are clearly specified these conflicts can be minimized. In this case, the goals should be stated in terms of deer populations and endangered species populations rather than fire return interval. It is important to acknowledge subjectivity upfront and not assume goals or values are objective when they are not. On the other hand legitimate conflicts do occur; for example, different species might inhabit early seral vs. late seral habitats and the decision to enhance habitat for one harms the other. In this case it is unlikely that the underlying issue is differing values, rather, the value here may be persistence of biodiversity and the underlying problem may be uncertainty in that it is not clear which species is in greater need.

■ *One problem faced by decision makers is that certain value conflicts play out as goal conflicts such as the maintenance of military readiness and conservation of endangered species. This happens because the choice of goals is based on values.*

12.2 Structuring Decisions to Incorporate Risk Analysis

Because of subjectivity and uncertainty, the risk assessor influences how decision problems are structured or specified. Risk can be thought of both in terms of the magnitude of harm or danger an outcome poses, and in terms of the probability that it might happen within a specific time frame (Burgman 2005). When people hold different values it follows that their perception of the risk posed by the consequences differs. Decisions under risk entail those situations where it is not certain what the state of the environment is but probabilities can be assigned to the likelihood of the occurrence of each different state. Risk is differentiated from uncertainty here because under uncertainty the probability of each state is not known.

■ *When people hold different values it follows that their perception of the risk posed by the consequences differs.*

The basic framework for decisions is a choice between two or more management actions, each of which will produce one of several outcomes depending on environmental circumstances or the state of the environment. One way uncertainty or risk enters the picture when insufficient information exists about the environmental circumstances to guarantee the outcome. This uncertainty may be in the form of not knowing the number of individuals in a population, not knowing the pattern of a fire within a plant community before it happens or not knowing what the precipitation will be in the coming year. This type of uncertainty is accounted for by assigning probabilities to the states (weather, numbers of individuals, etc.) of the environment.

Decision trees are one method that graphically represents a decision framework. They link chains of events from inception to all possible outcomes through a series of mutually exclusive choices (Figure 17). In decision trees the nodes or branch points are either events or decisions. The benefit of a tree is that it clearly and concisely shows the chain of events from beginning to end. Decision trees are a framework that consists of acts or management alternatives, states of the system, and outcomes or effects. Probabilities are typically assigned to each state such as the probability of wildfire. Values are then determined for the outcomes. It is important to remember that subjectivity is included both in the probabilities and values; since it is built into the framework it is not always obvious. Probabilities contain subjectivity because they are often the result of expert opinion and values because they represent preferences of the decision makers and stakeholders. In addition, even when a value is the result of a calculation, what goes into the calculation is a matter of choice. For example, the cost of lost habitat could be chosen to be represented by the cost to recreate it (if feasible). However, the cost of restoration alone ignores the temporal value of the habitat between the time of destruction and restoration. In addition this does not account for the increased risk of extinction over the time that the habitat is being restored. Decisions also involve balancing trade-offs that are not valued in the same currency. For example, the cost of management to reduce the risk of extinction and the value of increased military readiness cannot be reduced to a common currency.

In order to complete the analysis of this information the probabilities and values feed into utilities. Utility is a concept that takes into account the decision maker's preferences, and the chance or probability of a specific outcome. Utilities are subjective not only because they depend on an individual's preferences but because estimates of probabilities are often subjective (Burgman 2005). An example of a species-based utility could be the percentage increase in projected population size as a result of different management scenarios. In this case a population dynamical model for the species could estimate the future increase. The decision maker then chooses the set of actions that lead to the outcome with the highest utility.

The assumptions for using decision trees are that all relevant states are included, the states are mutually exclusive and not changing or interactive, and the probabilities and utilities are correct. Often however, these assumptions are violated. For example, fire and drought might be included as two states that could influence outcomes. The interaction between them may be, in fact is likely to be, more complex than a simple additive relationship. Also, states of the system are not always static and we cannot always predict how they will change.

Figure 17 shows a simple example of a decision tree with probabilities, values and utilities. This example evaluates the controversy over managing an endangered species as a single larger population or as more than one, but smaller, populations. The states in Figure 17 relate to catastrophic storm events. The probability of those events was calculated from historic weather records. To evaluate the options two models are needed one which estimates the population viability when storms occur and the other estimates the population viability when storms do not occur. The model parameters listed in Figure 18 were used.

The models were then run for two scenarios each: 1) a large population with a storm and without a storm; and 2) a small population with a storm and without a storm. The outcome, the probability of population survival, is simply one minus the probability of extinction predicted by the model. The utilities are the probability of a given state (a storm occurring or not) multiplied by the outcome. Then for each of the possible actions, the states, and the utilities are summed to give the maximum expected utility. In this case, given the likelihood of storm events and the vital rates of the populations, the management scenario maintaining two small populations has the highest maximum expected utility.

Figure 17. Decision tree for comparison of the value of a single large against two small populations.

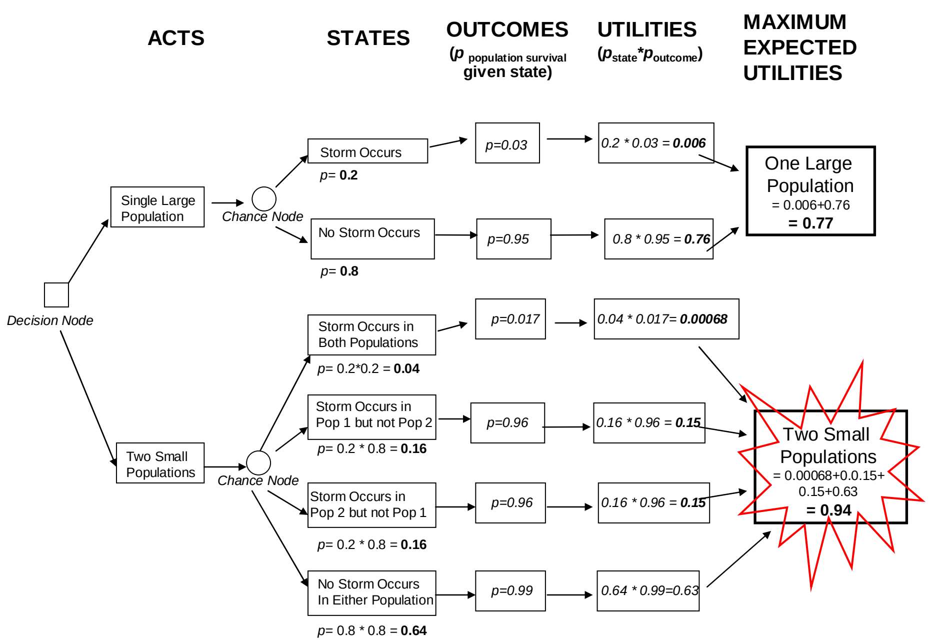


Figure 18. Model parameters for whooping crane population with and without storms.

Model parameter	Without storms	With storm
Growth rate	0.9	0.8
Survival rate	0.7	0.6
Std Dev of R	0.1	0.1
Density dependence	Exponential	Exponential

Uncertainty enters the decision process through the establishment of probabilities and values. This type of uncertainty derives from incomplete knowledge of the system. For example we may not know precisely the effect of storms. It can also be introduced through errors in the application of management. Perhaps we know with certainty the outcome of a specific management activity, however if it is implemented incorrectly the expected outcome might be a surprise. Probabilities cannot legitimately be assigned to these uncertainties because we do not even know the shape of the distributions. In some cases bounds can be assigned. However when even this cannot be done, information gap theory provides a method for determining which decision is the most robust to uncertainty. In other words, what decision will prevent unacceptable outcomes while maximizing the likelihood of a good outcome?

13.0 DECISIONS UNDER SEVERE UNCERTAINTY

■ *Particularly with endangered species, management decisions are often made under severe uncertainty (Regan et al. 2003).*

■ *When uncertainty is ignored, conservation and management decisions can have adverse unintended consequences*

Decision theory typically does not consider uncertainty, when estimating the probabilities assigned to the states of the system (for example the probability of a severe drought) or in the outcomes (how the population of an endangered species will respond to habitat restoration) and in the subsequent utilities. This can lead to significant underestimation of risk. Particularly with endangered species, management decisions are often made under severe uncertainty (Regan et al. 2003). This can be due to small sample size, short-term data sets, or simply a lack of information. For some migratory species, where they spend winters may not be known. When uncertainty is ignored, conservation and management decisions can have adverse unintended consequences (Halpern 2006; Regan et al. 2005).

There are several options to address severe uncertainty. One is to describe the probability distributions for the parameters and randomly select parameters from these distributions. However when uncertainty is severe it may not be possible to characterize the probability distributions for the parameters sufficiently to model them. A method that can be used when the probability distributions are poorly understood as long as the parameter values can be bounded is interval analysis. This involves setting bounds that include all possible parameter values (Regan et al. 2005). However, even this is not always possible. When the uncertainty cannot be described or bounded then information-gap theory (Ben-Haim 2001) can be used to evaluate the effect of uncertainty on reliability of a given decision. While standard decision theory seeks to maximize the

expected utility, info-gap analysis seeks to maximize the chance that an acceptable outcome is achieved. One benefit of the info-gap approach is that by determining what amount of uncertainty is acceptable before a decision would change it helps to avoid costs in reducing uncertainty of given parameters if that reduction in uncertainty would not change the decision (Halpern 2006). For example, the question might be asked: would additional surveys to improve fledgling mortality estimates change the management approach or not.

Information-gap theory requires a model which mathematically describes a process (such as population dynamics), a model that describes what is known about the uncertainty in the parameters in the process model, and a performance criterion (probability of extinction) which can be calculated using the process model. With these elements the sensitivity of a decision can be evaluated with respect to uncertainties in the parameters and the structure and functions used in the models. It provides the decision maker with information on the susceptibility of the decision to both unacceptable and highly favorable outcomes. This allows the decision maker to trade-off insurance against undesirable outcomes high performance. One can calculate the maximum level of uncertainty that guarantees that an expected utility is not less that a specified critical threshold (Halpern 2006; Regan et al. 2005).

14.0 SPATIAL DECISION SUPPORT SYSTEMS (SDSS)

SDSS are Decision Support Systems based on Geographic Information Systems designed to enable a flexible analysis of geographic information (Larson 2004). Rather than review the literature, our purpose here is to show how the methods discussed here can integrate with the SDSS in use by DoD for natural resources management issues. NatureServe is an SDSS for much of the Western Hemisphere. The DoD works with NatureServe to develop and analyze information. For example, a recent geographic analysis by NatureServe focused on imperiled species not yet listed under the ESA. This analysis documented the numbers and density of at-risk species on and near installations at the population level. It further documented which species were restricted to DoD lands and which regions of the country had the highest density of imperiled species. This kind of analysis helps DoD prioritize funding and target species which may not yet be regulated and potentially reduce the need to list certain species (NatureServe 2005).

This type of geographic analysis could be enhanced by a population dynamical analysis such as proposed here. The initial analysis might indicate which regions and which species to focus on but a multi-species analysis and plan based on population dynamics can better facilitate choices between management options, and in the case that one species must be favored over another, which species is the appropriate target.

15.0 ECOLOGICAL AND MANAGEMENT STATUS MEASURES (METRICS)

- *Metrics can be viewed from two perspectives. They serve to inform the leadership and the public about what is being accomplished and they serve to remind the organizational team about what is important.*

Within the DoD, measures that give the status of natural resources management and INRMP implementation are known as "metrics." These measures are particularly important in light of the requirement that in order for DoD to be able to use INRMPs in lieu of critical habitat designations under the ESA the plans must provide effective conservation benefits to listed species and be implemented. Metrics can be viewed from two perspectives. They serve to inform the leadership and the public about what is being accomplished and they serve to remind the organizational team about what is important. No single measure is generally sufficient to characterize both program implementation and effectiveness of individual program elements. The key is to select a relatively small set of measures that are critical to success. There typically are trade-offs between important aspects of the mission such as long-term investments and short-term returns (Kaplan and Norton 1992). This is also true of natural resources management.

The challenge for metrics is to portray accomplishments and progress towards overall DoD goals as well as specific INRMP goals in meaningful ways. It is important to communicate progress and accomplishments. For example, extinction probability is not a good metric due to high uncertainty, weak link to economic value, and insensitivity to short-term change. In contrast trends in population size, number of populations and amount of habitat could be informative metrics (Balmford 2003). Generally speaking there are two kinds of metrics: activity measures which mark actions that are taken and outcome measures which describe status of the biological elements. Activity measures tell us that we are implementing our plan; outcome measures report plan success. Both are important (TNC 2005). The lack of clear, explicit objectives has been noted to be significant problem in conservation (Sainsbury et al. 2000; Failing and Gregory 2003; Nicholson and Possingham 2007). In fact metrics can be particularly hard to develop when we do not even know the system that well.

The measures chosen to assess the ecological status of target species and communities need to describe success of the selected management strategies and the key processes. Metrics in this context are measures that provide an indication as to whether an organization's goals and objectives are being met and its vision thus implemented. In fact organizational strategy and vision should be at the heart of a metric system. Metrics are inextricably tied to decision making. Accountability (both fiscal and conservation outcome based) demands metrics (Hockings 2003).

- *The DoD has a two-tiered approach: 1) INRMPs contain goals, objectives and metrics, and 2) INRMPs are collectively evaluated to ensure implementation.*

The DoD has a two-tiered approach: 1) INRMPs contain goals, objectives and metrics, and 2) INRMPs are collectively evaluated to ensure implementation. The Sikes Act requires activity metrics including annual

reporting on the number of INRMPs and implementation in terms on expenditures. DoD policy (OSD Guidance "Implementation of SAIA Amendments: Updated Guidance 2002) requires each of the Services to report on measures of merit but leaves the details to each branch. Within DoD metrics function by stakeholder consensus rather than by a conceptual or quantitative model. The Navy has developed their metric system around 7 focus areas that include both activity and outcome metrics:

1) Ecosystem integrity
2) Fish and Wildlife Management and Public Use
3) INRMP Impact on the Installation Mission
4) INRMP Project Implementation
5) Listed Species and Critical Habitat
6) Partnership Effectiveness
7) Staffing Adequacy

For each of these areas there are a series of questions that the stakeholder group scores. The scores are tallied and weighted for each focus area resulting in an overall score of: 1) satisfactory, 2) areas of concern and 3) problem areas. These or similar focus areas will be used by the other Services, however they will each develop their own series of questions for their scorecards (P. Boice, pers. comm. 2008).

Models have potential to support the development of objectives that could be used in outcome metrics. Specifically they could be used to identify the most important factors, limiting factors and significant data gaps. For example, considering the focus areas listed above PVAs could be used in developing a number of the rating criteria including:

- *Models have potential to support the development of objectives that could be used in outcome metrics.*

1) Ecosystem Integrity
   - Population Trends of Indicator Species.
2) Assessment of Listed Species and Critical Habitat
   - Do INRMP projects contribute to recovery.
   - Do INRMP projects and programs benefit candidate species.
   - Are baseline surveys adequate.

The use of threatened and endangered species as focal species for ecosystem health metrics can be problematic depending on the amount of data available and the health of the particular population. In general the more specific the metrics the more utility they will have in informing adaptive management and defining effective conservation measures (Higgins et al. 2007). In using metrics to assess the success of a program we need to consider whether success in one metric or value can be offset by another. If this is acceptable, multi-criteria methods could be used to evaluate trade-offs between metrics. Typically some degree of compensation between metrics will be acceptable though for the same

reason that a single metric cannot be used to determine success, however failure in one area cannot always be offset by success in another.

Ecosystem health metrics need to consider the relationship between conservation and threats. Metrics address threats in that threats influence what is at risk and needs to be measured. It is important to distinguish between current and future threats. If metapopulations are likely to decrease, stochastic extinction might be a future threat but not a current one. In many cases we need to consider multiple threats. TNC explicitly recognizes the importance of overlap between the status of biodiversity, threats and conservation (Higgins et al. 2007).

- *There tends to be a bias in favor of quantitative data but it is not always more accurate or relevant than qualitative data. Both are subject to error and require interpretation (Hockings 2003).*

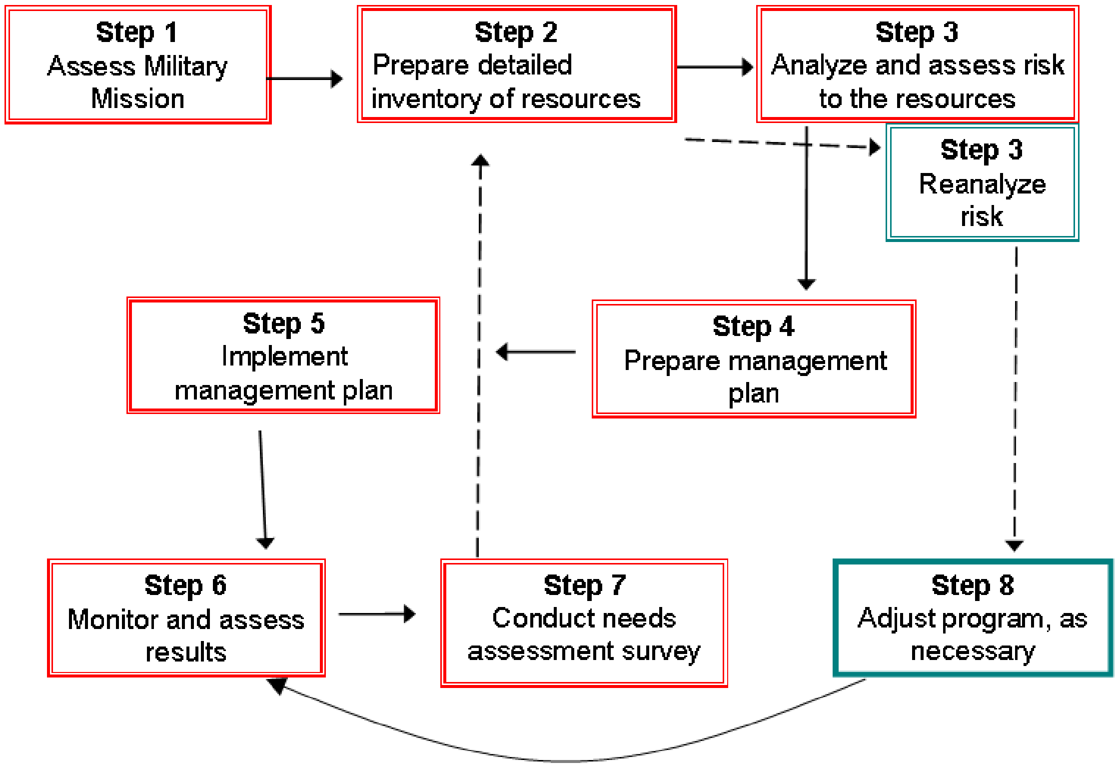
Finally it is not always easy to determine what constitutes "truth". For example, subjective responses of land managers are likely to be based on years of field experience and may better capture realities and complexities of ecosystems that short-term quantitative studies. Weiss and Bucuvalas (in Hockings 2003) argue that decision makers should apply both truth (accurate and believable versus absolutely true) and utility tests in judging evaluation information. There tends to be a bias in favor of quantitative data but it is not always more accurate or relevant than qualitative data. Both are subject to error and require interpretation (Hockings 2003).

It is clear that existing, fully protected landscapes are insufficient to protect biodiversity and that inclusion of landscapes with varying degrees of human impact are necessary to conserve biodiversity on earth. The issue then becomes how to manage and how to evaluate (metrics) to determine the degree of management effectiveness and the value towards regional and global conservation. DoD has taken on this responsibility but needs to ensure simultaneously that their primary mission of military preparedness is accomplished.

16.0 ADAPTIVE MANAGEMENT

Throughout this report adaptive management is recommended as a validation of management actions selected based on theoretical predictions and thus as a way to develop future management plans. In fact the INRMP process is an adaptive management process. Figure 19 shows a flow chart of the INRMP process as set out in DoD instruction 4715.3 which is clearly adaptive in approach with information obtained after plan implementation fed back into the planning process.

Figure 19. General Conservation Management from DoD Instruction 4715.3



Holling (1978) described adaptive management as a process where stakeholders integrate information on system behavior and project goals, generate alternative objectives, design policies to meet alternatives, systematically test and evaluate the policies and provide feedback to the decision maker. He proposed it as an alternative to "reactive management" with environmental factors integrated at the beginning of a process. Subsequently adaptive management has been characterized as either active or passive. Active adaptive management involves experimental hypothesis testing to evaluate specific policies or management actions and is designed to accelerate learning relative to passive adaptive management which relies on observation and not management experiments (Shea et al. 2002). Others define it more loosely as learning from the outcomes of ongoing management and that it is most effective when management options and policies are compared experimentally (Nyberg and Taylor 1995; Moir and Block 2001).

- *Adaptive management mitigates the risks of uncertainty by monitoring the system to test the predictions, validate decisions and change course as warranted.*

The central theme in adaptive management is uncertainty. Scientists and politicians view uncertainty from substantially different perspectives. Scientists see it as a system property and seek to understand and quantify uncertainty through experimental design and statistical tools. Politicians and the public on the other hand typically view uncertainty negatively and do not understand it as a normal system property (Clark 2002). As noted earlier uncertainty is what causes risk in the decision process. Adaptive

management mitigates the risks of uncertainty by monitoring the system to test the predictions, validate decisions and change course as warranted.

Holling (1978) defines adaptive management as more than trial and error, including hypotheses and experiments. Adaptive management is distinguished by implementation of alternative management scenarios using experimental designs designed to test for differences between treatments while moving forward with management decisions that must be made. For example if it is not clear which method, among several, of exotic plant control might be most effective but control actions must be initiated. The best methods are implemented in an experimental fashion. As information is developed adaptive management programs proceed with incremental iterative experimentation and feedback guiding management (Roe and Van Eeten 2001).

Adaptive management is highly recommended where processes and interactions are poorly understood (Meretsky 2000; Moir and Block 2001), which upon reflection includes most natural systems. It is clear however that it is not appropriate for all cases. Adaptive management is not appropriate where the risks of failure are unacceptable, for example with endangered species, methods cannot be tested which might drive species to extinction (Gunderson 1999). However, in the case of at-risk species often little is known and a course of action must be chosen, even status quo or complete protection is an action. Presumably the choice will be the one that gives the species the highest chance of survival given all the objectives of the decision maker but often the uncertainty surrounding the outcome of the management is high and monitoring the results and adjusting future management to reflect them is imperative.

Adaptive management has been used in a wide variety of natural resources management contexts including: fisheries management (Smith et al. 1998), forest management (Nyberg and Taylor 1995), endangered species management (Wilhere 2002), desertification on the Colorado Plateau (Clements 2004), weed control (Shea et al., 2002), ecological restoration (Thom 1997), and river management (Clark 2002). All of these authors cite uncertainty and system complexity as driving forces behind the adoption of adaptive management techniques.

A number of authors have concluded that adaptive management has been more important as an idea than a practical management tool citing institutional barriers as a key obstruction (Lee 1999; Clark 2002; Wilhere 2002). Fortunately or not, crisis is what often persuades decision makers to act. Adaptive management involves upfront costs and changes or potential changes in the way management is done and it appears that crisis in the form of lawsuits or regulatory pressure is often necessary to force adaptive management. Moir and Block (2001) conclude that failure to adequately design and carry out monitoring programs is the main problem.

They noted that high variation can obscure ecosystem responses, and most monitoring programs are not scaled to capture slow, long-term ecosystem processes. They specify a number of issues that prevent adaptive management from being adequately implemented including institutional barriers, inadequate funding for monitoring, data not available when decisions need to be made, and too many issues are tackled at one time.

One of its strength lies in the fact that it recognizes that uncertainty and complexity are undeniable, highly important components of natural ecosystems. Scientists and natural resource managers know that they are basing opinions and recommendations on best guesses. Therefore a management process that ensures consideration of that uncertainty, by design, has obvious appeal.

Adaptive management has a continued strong role improving in applied ecology; while start up costs due to monitoring may be high, substantial savings have been achieved through the adaptive approach, most demonstrably in weed control (Shea et al. 2002; Lawson et al. 2005). In addition Shea et al. (2002) showed that work done in a management context can make significant contributions to basic ecology.

A paradox in the successful application of adaptive management is that endangered species can prevent implementation because the laws around them are strict and perceived as inflexible. In fact, regulation is one of the crises that can force managers to adopt adaptive management. Endangered species also have the hallmarks of an appropriate target for adaptive management, namely a management crisis and high levels of uncertainty. Perhaps an effective use of adaptive management would be in cases where stakeholders come together to manage a species so that future listings and the costs they entail are not incurred.

■ *Endangered species also have the hallmarks of an appropriate target for adaptive management, namely a management crisis and high levels of uncertainty.*

Adaptive management programs offer an important opportunity to test mechanisms and processes at larger scales. Hobbs (2003) notes that complicated environmental problems require development of mechanistic understanding of ecosystem properties and processes and understanding how they function at a range of scales. Scientific investigation is often done at much smaller scales than management. At a basic level the methods used to apply a treatment to a square meter plot may not be the same as for a hundred acre management area so that in the case of weed control a different amount of herbicide delivered to a target plant may result. At a more subtle level, phenomena may be scale dependent and the mechanisms operating at one scale may not hold at a different scale. Once mechanisms are understood at small scales they can be used in basic or applied contexts to develop hypotheses to be tested at larger scales in adaptive management programs.

17.0 DISCUSSION AND CONCLUSIONS

The applied conservation focus of population viability analysis based on population dynamical models has shifted from attempts to develop generalities (Boyce 1992; Shaffer 1990) such as minimum viable population estimates to an emphasis on individual species (Reed et al. 2003; McCarthy et al. 2003). The lack of accuracy in model estimates has been widely debated in the literature and, as a result, modeling advocates recommend the use of relative rather than absolute results (Brook et al. 2002; Lindenmayer et al. 2003; McCarthy et al. 2003; McCarthy et al. 2001).

- *Relative model results have significant value to conservation decision making because, while they do not allow prediction of a specific number of years to extinction, they do support decisions by predicting which course of action from a set of alternatives will minimize the risk of extinction.*

Relative model results have significant value to conservation decision making because, while they do not allow prediction of a specific number of years to extinction, they do support decisions by predicting which course of action from a set of alternatives will minimize the risk of extinction. They can also be used to prioritize future data collection by identifying what vital rates model results are most sensitive to. PVAs, because they are rigorously structured, facilitate use of all the available data, are transparent and explicit about assumptions, easily repeatable, provide a way to quantify uncertainty or at least in cases where uncertainty is very high, evaluate the effect of uncertainty on the reliability of a given decision (Halpern 2006). One of the main values of PVA is in clarifying the problem (Brook et al. 2002). In fact, models are being used in conservation decision making on military lands (Kaiser et al. 2008).

The heart of modeling, sensitivity analyses, is probably conceptually close to what experts and managers do when they know their data are uncertain. They review what they know and come to a conclusion about what is most likely and then hopefully consider what would happen if they were wrong (alternative parameter values). The advantage of structured sensitivity analysis is that it is more consistent thorough and repeatable.

The simultaneous management of multiple species has become the imperative, not only because of the shift in conservation emphasis to focus on biodiversity (one of primary conservation goals of DoD), but because so many rare species have been officially listed under the federal ESA that trade-offs for habitat and funding have to be made among rare species. Trade-offs are central to balancing mandates to support military preparedness and multi-species conservation. They occur between both the objects of conservation such as species which can occupy the same habitat but require different seral stages and different land uses such as military training and conservation. The Sikes Act in its language governing INRMPs acknowledges trade-offs by saying that implementation of the plans shall result in no net loss military training opportunities. In that case, the word “net” implies that the decision process can include trade-offs between military training, conservation and economics.

Structured decision making explicitly facilitates selection among alternatives based on their consequences (Resnik 1987). Decision theory incorporates the values of the decision maker, the risk of the situation and the uncertainty in the outcome. Application of decision tools clarifies the logic and facts that support decisions. Not all decisions call for a methodical application of decision theory. In some cases the choices and outcomes are so limited or clear cut that it is rational to make a direct decision. An immediate need or danger might not allow time to run through a decision analysis, such as the case with a natural disaster like fire or flood. In complex cases where there is time to structure a decision multi-criteria decision methods can clarify complex problems with trade-offs that cannot be expressed in a common currency. The operative principle is to clarify the facts and trade-offs for the decision maker. Use the simplest method to do that.

The fact is that values enter decision making; they must. When the military decision makers make choices regarding their land management they first and foremost want to optimize the achievement of military readiness. They also want to comply with the Endangered Species Act and other laws and also perhaps have a need in some cases to go above and beyond to garner the support of the public or special interest groups such as NGOs whose mission is to conserve biological diversity. When they do that, it should be a rational decision process in which they get a benefit to their primary need for what they have to offer. For example, a NGO might offer to support a novel regulatory approach provided the result is more species' conservation benefit than would be achieved under the status quo. The military might agree to do more than the minimum legal requirement provided they get more mission flexibility than the status quo. This is what is known as a "win-win."

We face high uncertainty in the management of lands with endangered species. We must make decisions in the face of that uncertainty. The use of PVAs and decision support tools within an adaptive management framework gives us the best chance of acceptable outcomes. The anticipation is that this will take us beyond case-by-case conservative decision making based on fear of extinction and move us towards a series of decisions that are optimal for conservation of species and ecosystems. The degree of conservatism in a decision typically depends on attitudes to risk, the extent of uncertainty in the problem and the trade-off between costs and benefits and perceptions of these can be influenced by agency mission. While conservative decisions may be warranted on a case by case basis, the problem with a series of conservative decisions is that they are unlikely to optimize conservation programs because of the opportunity costs of investing in the target species versus other elements of conservation such as erosion control which may have more long term and or indirect benefits (Burgman 2005; Halpern 2006).

Methods to assess and aggregate impacts from multiple projects or sources within a few projects are needed. The analysis of cumulative effects is a natural progression of simple models for individual species and populations. Separate sets of vital rates need to be estimated for the spatial and temporal extent of the impacts to be considered in the analysis. A caution is that the estimation of additional sets of vital rates and standard deviations introduces more error into the modeling process. However, as stated previously, alternatives to models are not without error but rather it is obscured and the analysis is not as transparent, nor as likely to be internally consistent.

- *Models could be used to demonstrate that INRMPs are more cost effective, desirable and biologically relevant than managing species through critical habitat designations.*

Models could be used to demonstrate that INRMPs are more cost effective, desirable and biologically relevant than managing species through critical habitat designations. In fact the effectiveness of critical habitat designations in conserving endangered species has not been well documented (Hoekstra et al. 2002). The problem then becomes to demonstrate the effectiveness of INRMPs rather than comparing them to critical habitat designation. This can be done by using models to compare the relative value of alternative management and impact scenarios and then using the models and their results in a decision support framework.

The focus of this report has been management of threatened and endangered species. This perspective is narrow compared to the business of ecosystem management but it is a place to start. Models and decision theory can be used for other management questions but in order to successfully institutionalize use these tools, the scope should initially be limited.

- *Appendix A contains a series of recommendations for using models in natural resources planning and management.*

Widespread use of PVAs and decision theory in DoD land management will not necessarily be easy. Natural resource managers are often too busy to keep up on the latest literature and to stop and make a significant investment in learning a new process. There will undoubtedly be difficulties and lessons learned in the implementation of models and rigorously structured decision support. For the benefits to be realized the entire process must be implemented especially the adaptive management portion where the models are updated and plans revised. A modeling approach is likely to fail if land managers stop half way through because of resource constraints and try to fall back on model results without the revision and update process. Therefore, it is recommended that several test cases be undertaken to implement these methods. Appendix A contains a series of recommendations for using models in natural resources planning and management. In addition models are worth the effort. They serve as a framework for an adaptive management program, utilizing the data that are collected and ensuring that the data to be collected will most cost effectively contribute to improved predictions. They can be used to test if alternative scenarios which may be cheaper in dollars or training opportunity costs would be as effective.

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MULTI-SPECIES MANAGEMENT USING MODELING AND DECISION THEORY:

APPLICATIONS TO INTEGRATED NATURAL RESOURCES MANAGEMENT PLANNING

SUMMARY OF RECOMMENDATIONS

Use an explicit decision framework such as that shown in figure (A1) and include modeling in the INRMP development. Include model development as part of the INRMP process even if the data only allow for a coarse approach that identifies the most relevant data to collect. Ensure that the data specified to be collected by the INRMP will be used in a model and that it is the most relevant data to collect. The data collected should be prioritized such that it is the most meaningful data to improve model predictions that can be collected with the amount of funding available (Akçakaya 1999).

ANALYSIS AND PLANNING PHASE

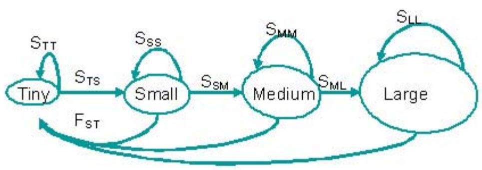
Step 1 - Collate data, identify problem, list options.

1. Establish objectives and goals.
    a. This can include long-term persistence of listed species, gaining additional beneficial information about the species, other natural resources management such as hunting, grazing timber extraction, no net loss of military training opportunities (or a specific increase in a given area) or fire management. Focus on multi-species management rather than single species. This list will almost certainly include competing goals and objectives and reveal value conflicts among stakeholders.
2. Identify individual species for which models will be developed. Recommend that this list prioritize federally listed species, and as resources are available, expand modeling efforts to other species.
3. Select one species to model. (One species would be a simple first step. You could also start with multiple species that perhaps have conflicting habitat needs or compete for other resources.)
    a. Establish a modeling team of various stakeholders.
    b. Refine objectives from step 1.1.a above. The objectives need to be clearly and explicitly stated in order to clarify trade-offs in step 7 below.
    c. Develop a range of management alternatives which may include for example specific habitat improvement practices, fire suppression or prescribed fire, varying land use (including military) scenarios where the intensity, frequency and seasonal timing may vary.

d. Conduct a literature review for current models on the species. Assemble the available data and model structure from both published and grey literature as well as expert opinion.

Step 2 - Determine (or modify) model structure.

Figure A1. Generic woody plant model structure. Ovals signify significant life stages and S are survival within and between stages and F represents fecundities.



1. Determine model structure or alternative model structures using the published and grey literature as well as expert opinion. Alternative model structures may be necessary in the case that so little is known about the species that the appropriate model structure is in doubt. Alternative structures can be evaluated in a sensitivity analysis.

Step 3 - Estimate or refine parameters.

Figure A2. Vital rate matrix. S represents survival parameters and F represents fecundities. The subscripts refer to the stages. Thus, STT is the survival rate for tiny individuals that remain in the tiny size class.

$$
\begin{array}{l c}
 & \begin{array}{cccc} \text{Tiny} & \text{Small} & \text{Medium} & \text{Large} \end{array} \\
\begin{array}{l} \text{Tiny} \\ \text{Small} \\ \text{Medium} \\ \text{Large} \end{array} &
\begin{bmatrix}
S_{TT} & F_{ST} & F_{MT} & F_{LT} \\
S_{TS} & S_{SS} & 0 & 0 \\
0 & S_{SM} & S_{MM} & 0 \\
0 & 0 & S_{ML} & S_{LL}
\end{bmatrix}
\end{array}
$$

1. Estimate parameters (figure A2) using the published and grey literature as well as expert opinion. This means developing models with existing information.

   a. Estimate parameters (and variance or upper and lower bounds) under current management scenarios. For example for a passerine this might include:

   - Survival rates of the life stages in the model
   - Fecundity of life stages in the model
   - % of juveniles that breed next season
   - **standard deviations on the above parameters**

b. Uncertainty can incorporated by estimating likely ranges of parameters and alternative model structures where precise information is not available, and comparing model results to get a range of predictions

c. Estimate separate sets of parameters and variance for all alternative management and land use scenarios to be evaluated. You may have to rely on expert opinion to develop this.

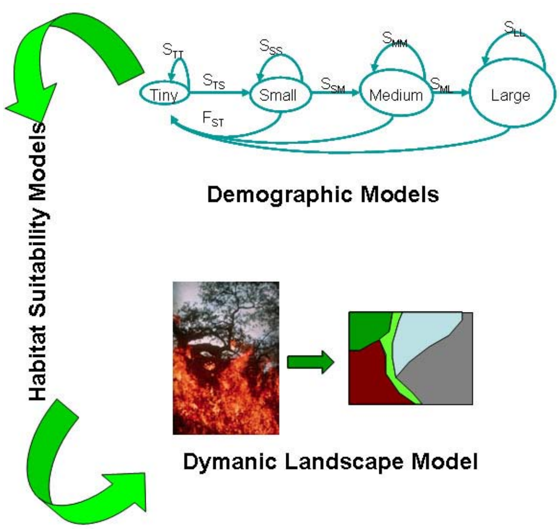
d. For a metapopulation model develop a map of occupied and suitable habitat.

e. Again if there are concerns about the uncertainty in the data or differing published and or expert opinion about parameter values, it should be established at this step. Different stakeholders could be concerned that others are too conservative or liberal in their parameter estimates. This is the place to document that and evaluate the significance of the concerns. It could be that given what is known, the influence of the parameter in question is small. Then during the sensitivity analysis the significance of the differing estimates can be evaluated.

Step 4 - Build (or improve) model.

1. For more advanced applications you can link the population dynamical model to a landscape model (such as Landis) so that your future predictions take into account anticipated habitat dynamics from factors such as fire and flood (figure A3).

Figure A3. Complex model linking demographic models to dynamic landscape models using a habitat suitability model.



2. Test or validate model predictions with data not used to construct the model. In cases where there is not enough data to validate the model at the time it is developed the model should be used to prioritize data collection efforts and then when those used to validate the model and inform future data collection in the adaptive management cycle.

Step 5 - Assess extinction risks and recovery chances.

1. Run model

2. Calculate confidence intervals if possible. If CI's cannot be calculated then only relative model results should be used (Fieberg and Ellner2001).

Step 6 - Perform sensitivity analysis.

Figure A4. Sensitivity analysis.

Parameter	Extinction Risk
Juvenile survival (S_J)	45%
(S_J) + 10%	24%
(S_J) – 10%	55%

1. On model structure if necessary.
2. On model parameters (Figure A4).
3. Repeat steps 3-5.
4. Use this information to develop future data collection priorities based on which parameters the model is most sensitive to, and where the most uncertainty exists. This will help ensure that the data you collect is the data you need the most.

Step 7 - Rank options and select optimal management plans.

The objective in this step is to clearly lay out all the facts and trade-offs for the decision maker. Use the simplest method to clarify facts and trade-offs that works.

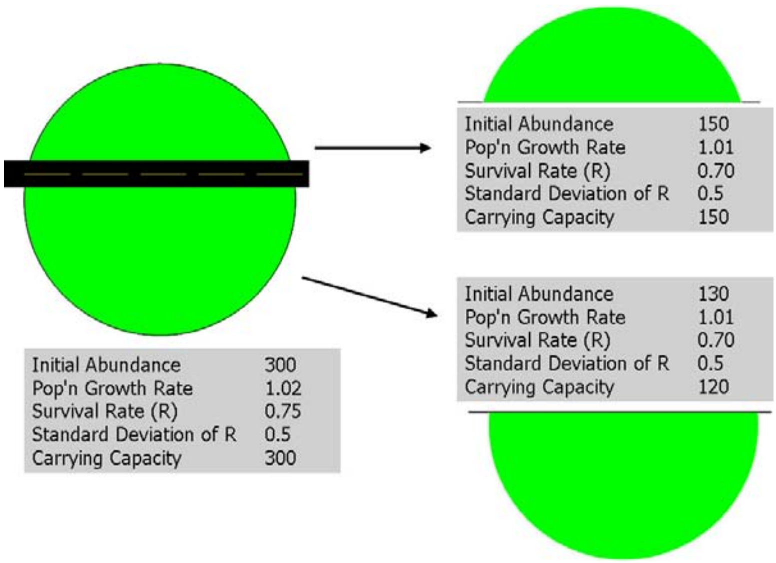
Figure A5. Trade-offs in habitat management between the California Gnatcatcher and the Stephens' kangaroo rat (SKR). Wildland fire can convert California Gnatcatcher habitat to SKR habitat.

1. Use info-gap analysis to evaluate the effect of uncertainty on the reliability of a given decision when uncertainty is high.
2. Use multi-criteria decision analysis methods where trade-offs exist between objectives that cannot be expressed in the same currency such as extinction risk and military training requirements.
    a. Select the best option from a group.

   b. Rank options.
   c. Graphically present results with two objectives, for example extinction risk and cost.
   d. Display pareto optimal solutions (solutions that are best for a given objective) when there is not a best solution.

3. Decision trees.
4. Generate tasks for INRMP.
5. This is also the step where impacts of alternatives can be evaluated for National Environmental Policy Act documents and Biological Assessments (figure 6).

Figure A6. A hypothetical impact analysis where new vital rates are estimated for habitat fragmented by a road.



ADAPTIVE MANAGEMENT PHASE

Step 8 - Implement the management plan.

Step 9 - Long-term Species Monitoring

Step 10 - Evaluate the monitoring data.

1. Test model accuracy using the new data collected.
2. Feed into ecosystem health metric.
3. Loop back to the beginning and revise the model structure if warranted.
4. Refine the parameter estimates as necessary and run back through the Decision Framework.
5. Revise INRMP tasks as necessary.
6. Continue with implementation and monitoring.

MULTI-SPECIES ANALYSIS

1. Develop single species models for each species and use one of the methods below.
    a. Use changes in relative risk of extinction for each species from alternative management/impact scenarios (Fox et.al. 2004).
    b. Develop weighted habitat suitability maps (Akçakaya 2000).

INCORPORATING CLIMATE CHANGE SCENARIOS (Keith et al. in press)

1. Develop single species models for each species.

2. Develop habitat suitability models for each species using the present climate as predictor variables.

    a. Use the HSM to predict habitat suitability under climate change.

    b. Interpolate habitat suitability maps for each time step of the population model.

3. Model population dynamics by integrating changing habitat suitability at each time step via the maps generated in step 2b.

ECOSYSTEM HEALTH METRIC

1. Use of Models for and ecosystem health Metric
2. Adopt a set of focal species to cover a range of habitats and ecosystem processes, develop single species models and evaluate management scenarios.

CUMULATIVE EFFECTS ANALYSES

1. Implement steps above to develop models addressing the spatial and temporal extent defined for the cumulative effects analysis in question.